Anaerobic treatment of domestic wastewater in subtropical regions

Lucas Seghezzo

Promotor:
Prof. dr. ir. Gatze Lettinga, Hoogleraar in de anaërobe zuiveringstechnologie en hergebruik van afvalstoffen

Co-promotoren:
Dr. ir. Grietje Zeeman, Universitair docent bij de sectie Milieutechnologie
Dr. Carlos Mario Cuevas, Universidad Nacional de Salta, Argentina

Samenstelling promotiecommissie:
Prof. dr. Henk Folmer (Wageningen Universiteit)
Dr. ir. Ronald Mulder (Paques B.V., Nederland)
Dr. ir. Bas van Vliet (Wageningen Universiteit)
Prof. dr. Fernando Fernández-Polanco (Universidad de Valladolid, España)

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Para Adriana, Natalia y Mateo

"...que hay algunos que se cansan en saber y averiguar cosas que, una vez sabidas y averiguadas, no importan un ardite al entendimiento ni a la memoria..."

(... because there are people who strive to know and find out things which, once known and found out, are not worth a straw to neither knowledge nor memory...)

Don Quijote de la Mancha

FOREWORD

This thesis ends an era in my life and will hopefully open the doors of my future career. However, the story of this thesis has many “beginnings”. The first beginning goes as far back as the mid 60s. I was born in one of the “Communities of the Ark” founded by the Italian philosopher and spiritual leader Lanza del Vasto (Shantidas), a close disciple of Mahatma Gandhi. The community was based on the principles of non-violence, religious tolerance, simplicity, self-sufficiency, social justice, and respect for nature. Three decades later, these principles oriented my research to a subject related to sustainable development.

The second beginning could be traced to the late 70s and early 80s. My mother used to tell me, with profound admiration, about the way her father, Augusto Raúl Cortazar, managed to obtain his Doctor degree at the University of Buenos Aires while working and taking care of the family. His perseverance, integrity, and outstanding scientific career have always been a source of inspiration to me. Somewhere in this period my father came out with the idea of building a little anaerobic digester to recycle our organic wastes and get energy for the family house in Villa Muñoz (Córdoba), former seat of the Community, and later in Vaqueros (Salta), where the family had moved in 1978. For a number of reasons this idea never materialized and I always felt that, eventually, it was going to be my task to build one.

The third beginning stretches during my university years in Salta (1983-1990). I wrote my first paper on anaerobic digestion in 1983 for a course given by the late Ennio Pontussi, a pioneer of environmental concerns in Salta. Somewhat later, in the alternative magazine “Mutantia”, I read the story of a Mexican “ecological house” called “Xochicalli” where an anaerobic digester played a central role. This house/project was owned/directed by Jesús Arias Chávez, who I had the privilege to meet later in the Netherlands. With the sketches of Xochicalli, I finally built my own 1-m^3 anaerobic digester with the help of some friends. It was quite a job to keep it running, and the energy rewards were very meager (just one or two kettles of hot water per day). In 1990, I defended my graduate thesis about the use of greenhouses to heat up anaerobic digesters. For some years, I kept doing research on anaerobic digestion and I started to run into more and more papers with the strange acronym “UASB” on them, written by someone called Gatze Lettinga and his co-workers.

The last and definitive beginning started by the end of 1993, when I received a letter signed by Prof. Lettinga *Himself* inviting me for a course on anaerobic treatment (I have to admit that it was a photocopy). The costs were enormous to me, but I decided that attending this course was more an investment than a luxury. While doing this course I learnt about the M.Sc. program at Wageningen University. Back in Salta, on a sunny Saturday morning, after going through all the brochures I brought with me, my wife Adriana finally suggested: *¿...y si nos vamos a Holanda?* (What if we go to the Netherlands?). And there we went, undeterred by the colossal amount of paperwork required, the entrance examination, the English test, and all the uncertainties ahead of us. While doing the M.Sc., I inevitably heard about the Ph.D. program!

Es larga la carretera/cuando uno mira atrás/vas cruzando las fronteras/sin darte cuenta quizás. (The highway is long/when you look back/you cross frontiers/without noticing)[1].

[1] Charly García (1972)

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I am very grateful to many people who supported me in writing this thesis.

First of all, I would like to thank my "promotor", Prof. Dr. ir. Gatze Lettinga, for his inspiring and motivating guidance throughout this research. His passion for anaerobic wastewater treatment and his commitment to sustainable development and global justice encouraged me to start and go on with the long and painstaking process of doing a Ph.D. thesis on this subject. Gatze: *we shall overcome*![2] because *The line it is drawn/the curse it is cast/the slow one now/will later be fast/as the present now/will later be past/the order is/rapidly fadin'./And the first one now/will later be last/for the times they are a-changin'*.[3]

I would like to express my sincere gratitude to my "co-promotoren", Dr. Grietje Zeeman and Dr. Carlos M. Cuevas, who managed to supervise my work in a framework of friendship, freedom, and openness. Without their support and helpful comments and criticisms of the typescript at various stages in its development, this thesis would never have been completed.

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A number of people have helped make this thesis better than it might have been, from the very beginning, and even before. I am especially grateful to Mario Kato (who suggested me to work first

[2] Lyrics derived from Charles Tindley's gospel song "I'll Overcome Some Day" (1900)
[3] Bob Dylan (1963)

with pre-settled sewage), Pedro Córdoba (who introduced me into the world of chemical oxygen demand and volatile fatty acids), Nidal Mahmoud (with whom I learnt how to perform specific methanogenic activity tests), Elías Razo Flores, Jules van Lier, Bert Hamelers (whose desk I invaded for a while), Wendy Sanders, Luis Cardón, Irene Upton, Renato Leitão, and several anonymous referees, each of whom has provided constructive criticisms of earlier versions of one or more of the chapters. I am thus most grateful to all of them for making this text possible.

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All my life I will be deeply indebted to my parents Isabel and Ricardo, my family, my family in law, and my friends in Argentina and elsewhere, for their love, care, emotional support, and guidance. Ya volveremos, no se tomen todo el mate!

My greatest thanks are to my wife, Adriana, for her love, companionship, and integrity, and to our lovely children, Natalia and Mateo, simply for their wonderful existence. You make everything worthwhile. Mis amores: ustedes son mi vida!

ABSTRACT

Seghezzo, L. (2004), *Anaerobic treatment of domestic wastewater in subtropical regions*, Ph.D. thesis, Wageningen University, Wageningen, the Netherlands.

In this thesis, the use of upflow anaerobic sludge bed (UASB) reactors for sewage treatment was studied in the city of Salta, northwestern Argentina. The climate in this region can be defined as subtropical with a dry season. Mean ambient temperature in the city is 16.5ºC. Mean sewage temperature during the experiments was 23.0°C (monthly minimum: 17.2°C; daily minimum: 12.6°C). A literature review on the use of upflow reactors for sewage treatment was performed, and a brief description of laboratory, pilot-scale, and full-scale applications from all over the world is presented.

Experiments were performed in two pilot plants. The first pilot plant was a UASB reactor installed after a conventional full-scale sedimentation tank (settler). The second pilot plant was a two-stage UASB system with posttreatment in five waste stabilization ponds (WSP) in series. In the first pilot plant, chemical oxygen demand (COD) removal efficiencies up to 84% in total COD and 92% in suspended COD were achieved at a mean hydraulic retention time (HRT) of 2 h in the settler and 5.6 h in the reactor, equivalent to an upflow velocity (V_{up}) of 0.71 m/h. A granular sludge bed developed in the UASB reactor probably due to the low concentration of suspended solids (SS) and COD in the influent and an adequate combination of HRT and V_{up}. Some of the granules were surprisingly big (up to 5 cm in diameter). The system studied was highly robust and efficient, it consistently delivered a final effluent in compliance with discharge standards for COD and SS, and it produced a small amount of well-stabilized sludge. In the two-step UASB system of the second pilot plant, COD removal efficiencies up to 89% were obtained at mean HRTs of 6.4 + 5.6 h (V_{up} = 0.62 + 0.70 m/h), with 83 and 36% removal in the first and second steps, respectively. The effluent concentration from the two-step UASB system was similar to that obtained from the first pilot plant. The performance of this system was not affected during the coldest period of the year, which usually lasts about three months. The anaerobic sludge from both reactors showed good stability, especially in summer time, and could be directly disposed of without further treatment. After the posttreatment in WSP, the effluent also complied with discharge standards for pathogenic microorganisms. It was finally concluded that a single-stage UASB reactor followed by a series of WSP could be a very efficient, reliable, compact, and simple system for the treatment of raw sewage in subtropical regions like Salta.

Finally, a comparative assessment of the sustainability of three technological options for sewage treatment was performed, in terms of a series of technical, environmental, social, and economic criteria and indicators. In this preliminary assessment it was found that, under local conditions, a system UASB + WSP was more sustainable than (a) an aerobic high-rate treatment system based on trickling filters, and (b) a system of conventional WSP. The assessment method used (a multi-criteria weighted-scale matrix) was simple to perform and sensitive enough to detect differences in sustainability between the options compared. A representative panel of local stakeholders must perform the actual assessment in a transparent and participatory way before any concrete policy decision is taken. The final plead of this thesis is that sustainable development will only be achieved through a fully democratic way of decision-making that can go beyond political and economic motivations and that may be able to solve environmental problems and social injustices.

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CHAPTER 1

General Introduction

ABSTRACT

The anaerobic treatment process is increasingly recognized as the core method of an advanced technology for environmental protection and resource preservation and it represents, combined with other proper methods, a sustainable and appropriate wastewater treatment system for developing countries. Anaerobic treatment of sewage is attracting more and more the attention of sanitary engineers and decision-makers. It is being used successfully in tropical countries, and there are some encouraging results from subtropical and temperate regions. In this chapter, the main characteristics of anaerobic sewage treatment are summarized, with special emphasis on the upflow anaerobic sludge blanket (UASB) reactor. The application of the UASB process to the direct treatment of sewage is reviewed, with examples from Europe, Asia, and the Americas. The UASB reactor appears today as a robust technology that is by far the most widely used high-rate anaerobic process for sewage treatment.

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INTRODUCTION

Sewage is the main point-source pollutant on a global scale (Gijzen, 2002). Between 90 and 95% of the sewage produced in the world is released into the environment without any treatment (Bartone *et al.*, 1994, according to Elliot, 1999; Niemczynowics, 1997). This figure includes countries as the Netherlands, where 97% of the sewage is currently being treated (Mels, 2001). On the other hand, virtually 100% of the wastewater produced in households from most of the cities and towns in some developing countries is commonly discharged untreated in water bodies like rivers and lakes, with immediate and sometimes disastrous effects on public health and the quality of the environment. Simple, affordable, and efficient sewage treatment systems are urgently needed in developing countries because most of the conventional technologies currently in use in industrialized nations are too expensive and complex (Grau, 1996). Appropriate and sustainable sewage treatment technologies will help to preserve biodiversity and maintain healthy (freshwater) ecosystems, in order to provide clean water, flood control, abundant fisheries, and other services of vital interest to human societies. Among the different treatment systems now available worldwide, the anaerobic process is attracting more and more the attention of sanitary engineers and decision-makers. It is being used successfully in tropical countries, and there are encouraging results from subtropical and temperate regions. In this chapter, the main characteristics of anaerobic sewage treatment are summarized, with special emphasis on the upflow anaerobic sludge bed (or blanket) (UASB) reactor developed in the early 70s by Lettinga and coworkers (Lettinga *et al*, 1980; Lettinga and Vinken, 1980). The application of the UASB process to the direct treatment of sewage is reviewed, with examples from Europe, Asia, and the Americas.

THE PROCESS OF ANAEROBIC DIGESTION

The anaerobic degradation of complex, particulate organic material has been described as a multistep process of series and parallel reactions (Gujer and Zehnder, 1983; Pavlostathis and Giraldo-Gomez, 1991). Gujer and Zehnder (1983) identified six distinct processes in an anaerobic digester. With the addition of the homoacetogenesis as a subprocess on its own (Pavlostathis and Giraldo-Gomez, 1991), seven subprocesses are now recognized (**Table 1**). Several groups of bacteria catalyze the reactions taking place during the process of anaerobic digestion: (1) fermentative bacteria, (2) hydrogen-producing acetogenic bacteria, (3) hydrogen-consuming acetogenic bacteria, (4) carbon dioxide-reducing methanogens, and (5) aceticlastic methanogens. A scheme of the process of anaerobic digestion is presented in **Figure 1**.

Table 1. Subprocesses in the anaerobic digestion of organic polymeric materials.

1.	Hydrolysis of complex, particulate organic materials (proteins, carbohydrates, and lipids)
2.	Fermentation of amino acids and sugars
3.	Anaerobic oxidation of long-chain fatty acids and alcohols
4.	Anaerobic oxidation of intermediary products such as short-chain fatty acids (except acetate)
5.	Acetate production from carbon dioxide and hydrogen (homoacetogenesis)
6.	Conversion of acetate to methane (aceticlastic methanogenesis)
7.	Methane production by reduction of carbon dioxide by hydrogen

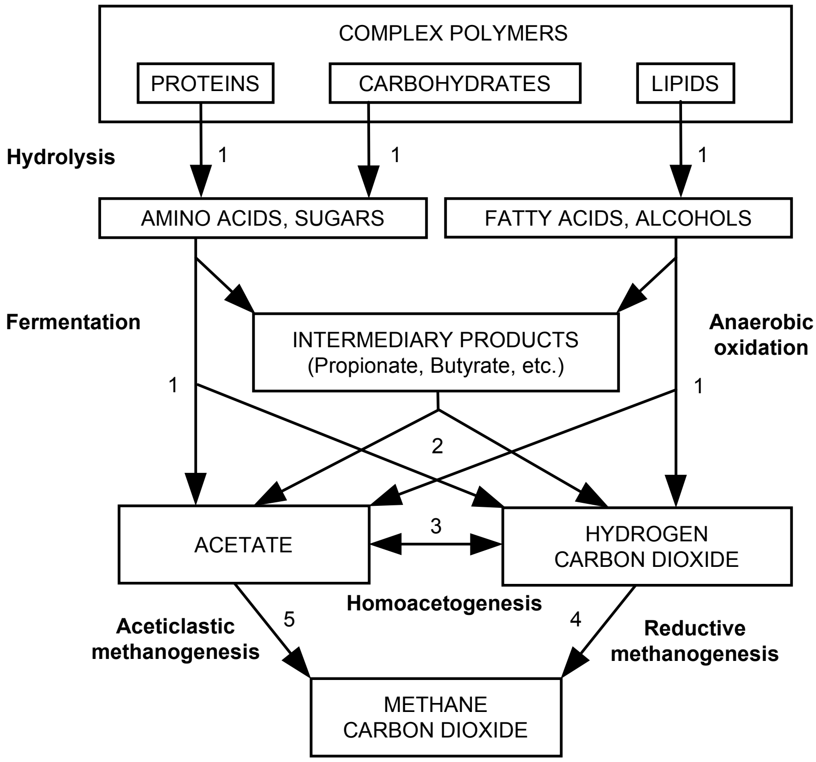

Figure 1. Anaerobic digestion of organic polymeric materials. Subprocesses in boldface. Numbers refer to the bacterial groups involved (see text).

Acetate is the major intermediate in the bioconversion of organic matter to methane and CO_2. About 70% of the total methane produced in anaerobic digestion originates from acetate (Gujer and Zehnder, 1983). Thus, the production of methane from acetate is an important step in the anaerobic digestion process. It is accepted that methanogenesis from acetate can proceed well from pH 6 to pH 8. If the pH drops below 6, methanogenesis from acetate will be inhibited and acetate will accumulate. Low pH and acetate accumulation, especially as undissociated acetic acid, further inhibit the degradation of fatty acids (particularly propionate) (Fukuzaki *et al.*, 1990). Since the conversion of acetate to methane is generally accepted as the rate-limiting step in methanogenesis from liquid waste (soluble substrates), the kinetics of the acetate-utilizing methanogens have been extensively studied (Gujer and Zehnder, 1983; Fukuzaki *et al.*, 1990; Lepistö and Rintala, 1995). Thereupon, acetate appears as a handy substrate to evaluate the effect of operational parameters on the kinetics of anaerobic sludges.

Some factors affecting anaerobic degradation

Temperature

The temperature dependence of the biological reaction-rate constants is very important in assessing the overall efficiency of a biological treatment process. Temperature not only influences the metabolic activities of the microbial population but also has a profound effect on such factors as gas-transfer rates and the settling characteristics of the biological solids. The efficiency of the anaerobic process is highly dependent on the reactor temperature (Bogte *et al.*, 1993; van Haandel and Lettinga, 1994). The optimum range for mesophilic digestion is between 30 and 40°C, and for temperatures below the optimum range the digestion rate decreases by about 11% for each °C temperature decrease, according to the Arrhenius expression. Temperature affects not only the rate of the process, but also the final degradation extent. At low temperatures, more organic matter will remain undegraded at a given hydraulic retention time (HRT) due to slow hydrolysis of volatile solids. However, as long as the solids can be retained in the anaerobic reactor, they are removed from the liquid phase. The entrapped solids have been successfully degraded in a separate heated digester (Mahmoud, 2002). Bogte *et al.* (1993) found evidence of accumulation of biodegradable solids during wintertime and degradation during summer time when operating small-scale UASB reactors for on-site sewage treatment in the Netherlands. When operating a hydrolysis upflow sludge bed (HUSB) reactor, Wang (1994) found no relationship between temperature and suspended solids (SS), chemical oxygen demand (COD), and biological oxygen demand (BOD) removal efficiencies. This fact was attributed to the large amount of sludge retained in the reactor (the average sludge concentration exceeded 15 g/L for the whole reactor), and to the accumulation of sludge in the reactor at low temperatures. The effect of temperature may be different for the various physical, chemical, and biological processes taking place in the reactor, and this fact must be taken into consideration when modeling the system.

pH

The value and stability of the pH in an anaerobic reactor is extremely important because methanogenesis only proceeds at a high rate when the pH is maintained in the neutral range (6.3 to 7.8) (van Haandel and Lettinga, 1994). When treating a complex wastewater like domestic sewage, pH is usually in the optimum range without the need for chemical addition, due to the buffering capacity of the most important acid-base system in an anaerobic digester: the carbonate system (van Haandel and Lettinga, 1994). On-line monitoring of the pH can be helpful in detecting system failure (Graef and Andrews, 1974). In some cases, it may lead to a reduction of the need for external alkali addition without significant deterioration of the final effluent quality (Romli *et al.*, 1994).

Particle deposition

Waters and wastewaters often contain significant amounts of colloidal and particulate matter in addition to soluble substances. Colloidal particles play an important role in the distribution of pollutants in natural aquatic systems because they may adsorb significant quantities of both inorganic and organic substances due to their large surface area relative to their mass (Filella *et al.*, 1993). Physical properties of sludge aggregates such as size, density, porosity, as well as the terminal settling velocity, have an important impact on the efficiency of solid/liquid separation operations (Námer and Ganczarczyk, 1993). Bouwer (1987) theoretically studied processes that affect the transport and fate of solid particles in biofilm systems. According to him, particle deposition in biofilm systems can be conveniently separated into two steps: 1) transport from the bulk liquid to the biofilm/liquid interface, and 2) attachment to the biofilm surface. The first step is mainly a physical process, while the second is primarily chemical.

Also, detachment of particles due to hydraulic shear forces or changing surface conditions may occur. Diffusion is much slower for particles than for a soluble substrate. In addition, a particle deposited on a biofilm will not likely penetrate far into the fixed biomass because it will be filtered in the outside layer of the biofilm. This indicates that the kinetics for removal of particulate BOD can be considerably slower than the kinetics for soluble substrates. The actual retention time of particulate organic matter can be considerably increased over the HRT if sedimentation or filtration retains the particles. When particular organic matter is retained, its concentration builds up and the removal kinetics expressed per unit of reactor volume, increase. Thus, HRTs shorter than required for a low-load regime can provide a long solids (or sludge) retention time (SRT) and performance equivalent to a low load when the solids are retained by sedimentation or filtration (Rittmann and Baskin, 1985). This is particularly advantageous for upflow fixed-bed operation where biological treatment and clarification are combined in one reactor. Column experiments with raw sewage under anaerobic conditions demonstrated that solids settling in the reactor produced a better effluent total BOD and more methane gas. Approximately 68% COD removal and 63% COD stabilization were achieved when sewage was treated at 23°C in an upflow anaerobic filter that allowed sedimentation of supra-colloidal particles. When sedimentation was prevented, COD removal and stabilization were only 45 and 43%, respectively. The difference in removals was attributable mainly to suspended particles (Rittmann and Baskin, 1985).

In essence the same as particle deposition, flocculation is a process during which particles suspended in water aggregate into larger masses so that they may be removed from the water in subsequent treatment processes, particularly by sedimentation (Bohle, 1993). Two-particle collisions are considered to occur by Brownian motion, fluid motion (shear), and differential sedimentation. Hydrodynamics and van der Waals attractions also play a role on particle collisions (Lawler, 1993). Forces between particles in water become especially important when the particles are in the colloidal size range (less than a few μm) (Gregory, 1993). Modeling of flocculation phenomena is not yet well developed and flocculator design and operation are still largely based on empiricism. However, the possibility of more rapid and accurate measurements of the particle size distribution in the influent will certainly favor the utilization of mathematical modeling to predict flocculation in wastewater treatment (Bohle, 1993). The role of flocculation on sludge accumulation processes in upflow anaerobic reactors has not been explored yet. Rebhun and Lurie (1993) give a review of the different methods of coagulation-flocculation for organic matter removal.

Mixing

The dynamics of the liquid flow and the sludge movements in an upflow reactor are interdependent and influence the performance of the process (Heertjes and van der Meer, 1978). The upflow velocity and the rising biogas bubbles are the main factors influencing the fluid flow and the mixing pattern within the reactor. The most effective contact between sludge and substrate would be obtained by homogeneous distribution of the influent over the bottom of the reactor and by steady biogas production. An idea of the mixing pattern in the reactor is necessary to understand the system and simulate it in mathematical models. A model for the fluid flow in the system can be built up based on stimulus-response techniques. The mixing pattern in the reactors can be determined using residence time distribution tests (Murphy, 1971; Levenspiel, 1972). The residence time distribution method is based on the age distribution of fluid leaving a vessel, or E distribution. Elements of fluid taking different routes through the reactor may require different lengths of time to pass through the vessel. The distribution of these times for the stream of fluid leaving the vessel is the E distribution, or the residence time distribution of the fluid. The response of the system to an impulse of tracer is recorded and the normalized effluent tracer concentration is represented as a function of time. UASB reactors have been usually considered as divided in three compartments: the sludge bed and the sludge blanket, both perfectly mixed with respect to the liquid phase, and a plug-flow region, assumed to describe the internal settler (van der Meer, 1979; Bolle *et al.*, 1986). Some bypassing and return flows, as well as dead space were also used to model the fluid regime more accurately (Heertjes and van der Meer, 1978). According to the height of the reactor, some differences have been detected in the number of mixers necessary to model the system (Heertjes *et al.*, 1982).

Kinetics of anaerobic digestion

Process kinetics has been used for the mathematical description of both aerobic and anaerobic biological treatment processes. The understanding of process kinetics is essential for the rational design and operation of biological waste treatment systems and for predicting system stability, effluent quality, and waste stabilization (Chen and Hashimoto, 1980). Process kinetics plays an important role in the development and operation of anaerobic systems. Based on the biochemistry and microbiology of the anaerobic process, kinetics provides a rational basis for process analysis, control and design. Sound knowledge on kinetics leads to optimization of performance, a more stable operation, and a better control of the process (Pavlostathis and Giraldo-Gomez, 1991). The kinetic description of anaerobic degradation of complex organic matter was generally accomplished through the so-called rate-limiting step approach. Hydrolysis was found to have a paramount importance in the overall process kinetics, even in cases where acidogenesis or methanogenesis were considered to be limiting steps (Pavlostathis and Gossett, 1988; Pavlostathis and Giraldo-Gomez, 1991). It is quite clear that the type of waste being digested (soluble, particulate, etc.) dictates which steps need to be considered. In anaerobic digestion, the rate-limiting step in the overall process is related to the nature of the substrate, process configuration, temperature, and loading rate (Speece, 1983). Raw cellulosics such as straw, corn stover, peat and wood are mainly limited in the hydrolysis step by the lignin sheath surrounding the cellulose. The recalcitrance of lignin to anaerobic biodegradation severely limits the hydrolysis rate of raw cellulosics. Grease and lipid biodegradation may be rate controlling in some industrial wastewasters. Food-processing industrial wastewaters are often high in starch and sugar content because of cooking operations. These simple organics are rapidly fermented to volatile acids. Consequently, the rate-controlling step is the conversion of the volatile acids to methane. Since complex wastewaters containing organics have a continuous range of degradation rates, at low loading rates, the rate-controlling step may be acid formation, as is evidenced by low volatile acids concentrations. But as the loading rate increases, the methanogenesis stage may gradually become the rate-controlling step, as evidenced by an accumulation of volatile acids (Speece, 1983).

Hydrolysis was found to be the rate-limiting step in the degradation of wastewaters high in volatile suspended solids (de Baere *et al.*, 1984). Hydrolysis rate in anaerobic systems is normally described as a first-order process with respect to the concentration of degradable particulate organic matter (Pavlostathis and Gossett, 1985; Pavlostathis and Giraldo-Gomez, 1991). However, it was reported that for some types of organic wastes like primary sewage sludge, none of its components (carbohydrates, proteins and lipids) was hydrolyzed following first order kinetics (Miron *et al.*, 2000). According to Hobson (1987), hydrolysis would be best described based on substrate available surface, instead of substrate concentration. Sanders (2001) verified the so-called surface based kinetics model in several batch experiments with different kinds of particulate starch as substrate. Her results show that, unlike the first-order hydrolysis constant, the surface-based hydrolysis constant was not affected by the particle size distribution of the substrate. Therefore, first-order hydrolysis constants from literature can only be extrapolated to substrates with the same particle size distribution.

Preliminary conversion mechanisms such as cell death and lysis are the first steps in the process of rendering viable biological sludge organisms to available substrate and can also be included in a kinetic model of the anaerobic digestion of biological sludges. However, these processes have been found not to be rate limiting (Pavlostathis and Gossett, 1985). Only about 10% of the waste activated sludge from commercial-scale activated sludge units may be viable cells, and lysis of viable cells is not expected to affect significantly the waste activated sludge digestibility (Ghosh, 1991). Even if cell death and lysis were an important factor in the overall kinetics of anaerobic degradation, these processes are not applicable in the case of treatment of raw domestic sewage, and therefore hydrolysis may be considered as the only possible rate-limiting step in this case (Sanders, 2001).

Biological treatment processes have been successfully described by the theory of continuous cultivation of microorganisms (Monod, 1950; Novick and Szilard, 1950; Herbert *et al.*, 1956). Continuous culture is usually defined as the continuous addition of substrate to a reactor containing microorganisms with withdrawal of the reactor contents in the same manner (Andrews, 1971). Under steady state, the specific growth rate is numerically equal to the reciprocal of the HRT. Monod (1950), among others, showed that bacterial growth rate was a function not only of organism concentration but also of some limiting nutrient concentration. He described this relationship with a hyperbolic function similar to the Michaelis-Menten equation used for describing enzyme-substrate interaction. Conversion rates during anaerobic treatment of soluble substrates are generally described by Monod kinetics (Pavlostathis and Giraldo-Gómez, 1991; van Haandel and Lettinga, 1994). According to Monod kinetics, the growth rate of microorganisms can be defined as follows:

$$r_x = \frac{dX}{dt} = \mu \cdot X = \mu_{\max} \cdot \frac{S}{K_s + S} \cdot X \qquad (1)$$

where r_x = growth rate expressed as volatile suspended solids (VSS) produced per unit volume per unit time (gVSS/L.d); X = microorganism concentration (gVSS/L); S = substrate concentration (gCOD/L); μ = specific growth rate of micro-organisms (d^{-1}); μ_{max} = maximum specific growth rate (d^{-1}); and K_s = Monod (half-saturation) constant (gCOD/L). The parameter μ is the relative increase of mass per time unit, while K_s represents the concentration of substrate at which the specific growth rate is half the maximum. There are two limit cases that can be used to simplify the mathematical analysis of the Monod equation:

$$\text{(a) When } S >> K_s,\ \mu = \mu_{\max} \qquad (2)$$

$$\text{(b) When } S << K_s,\ \mu = \mu_{\max} \cdot \frac{S}{K_s} \qquad (3)$$

At high substrate concentration (equation 2) the Monod kinetics may be approached by zero-order kinetics and the growth rate becomes independent of the substrate concentration. At low substrate concentrations (equation 3), the growth rate is proportional to the substrate concentration, which is characteristic of a first-order process. For intermediate concentrations, the growth rate is between zero and first-order with respect to the substrate concentration.

In addition to growth, other important process playing a role in anaerobic treatment is the decay rate of micro-organisms, which can be expressed by the first order equation

$$\frac{dX}{dt} = -b \cdot X \qquad (4)$$

where b = death rate constant (d^{-1}). To account for the effect of decay, which can be due to predation, cell death and lysis, and other factors, the net growth rate equation can be corrected as follows:

$$r_x = \frac{dX}{dt} = (\mu - b) \cdot X \tag{5}$$

The yield coefficient can be defined as the amount of biomass that is produced from a certain amount of substrate:

$$Y = \frac{r_x}{r_s} \tag{6}$$

where Y = yield coefficient (gVSS/gCOD); r_s = dS/dt, is the rate of substrate consumption (gCOD/L.d). The yield is expressed as the mass of biomass formed per mass of substrate consumed. Therefore, combining equations 1 and 6:

$$r_s = \frac{\mu \cdot X}{Y} = \frac{X}{Y} \cdot \mu_{\max} \frac{S}{K_s + S} \tag{7}$$

The ratio between μ and Y is often referred to as the specific activity (A) of the micro-organism involved, where the units are mass of substrate per mass of biomass per unit time, usually gCOD/gVSS.d. The maximum specific activity will then be

$$A_{\max} = \frac{\mu_{\max}}{Y} \tag{8}$$

The objective of any treatment system is to eliminate as much COD as possible from the wastewater. In some cases, the reactor substrate concentration is very low, and the effluent concentration could be determined by the value of K_S. Thus, sludge with high affinity for the substrate is required (low K_S). When methanogenesis takes place from acetate, K_s values encountered range from 18 to 30 mg/L for species of the genus *Methanotrix*, and from 257 to 300 mg/L for *Methanosarcina* species (Kato, 1994). These values vary from 15 to 930 mgCOD/L for mixed cultures and from 11 to 421 mgCOD/L for acetate-utilizing sludge in anaerobic treatment processes (Pavlostathis and Giraldo-Gomez, 1991). We have to distinguish between apparent and intrinsic K_s values. The former is measured in practical conditions, while the latter, which refers to the transport of substrate into dispersed bacterial cells in suspension, has to be determined in completely mixed systems, where transfer limitations across the cell membrane are assumed to be negligible.

Mathematical modeling of the anaerobic fermentation of organic matter can be an important tool in gaining an understanding of the process kinetics, in creating and implementing rational designs of biological waste treatment systems and in predicting system stability, effluent quality, and waste stabilization (Chen *et al.*, 1988). Although anaerobic processes involve dynamic changes during start-up, feeding, daily maintenance, and process failure, most mathematical models have been limited to the prediction of steady-state anaerobic processes (Lawrence and McCarty, 1969; Chen and Hashimoto, 1978; 1980). Dynamic modeling of anaerobic fermentation of industrial and animal wastes has been initially studied by Andrews (1971), Graef and Andrews (1974), and Hill and Barth (1977). Since anaerobic processes are very complex, quantitative simulations have, in general, been very complicated and involved many assumptions to achieve quantitative predictions. Organic and inorganic materials are involved in the process, as well as liquid and gaseous phases. However, mathematical modeling can be used to interface the fundamental characteristics of the processes; then, with the aid of computer simulation, an understanding of the overall operation can be developed (Hill and Barth, 1977). Simultaneous physical, chemical and biological processes taking place in an anaerobic reactor have been taken into account in "The IWA Anaerobic Digestion Model Nº1 (ADM1)", a comprehensive mathematical model of the anaerobic digestion process developed by an IWA (International Water Association) task group (Batstone *et al.*, 2001).

SEWAGE

The term "sewage" refers to the wastewater produced by a community, which may originate from three different sources: a) *domestic wastewater*, generated from bathrooms and toilets, and activities such as cooking, washing, etc., b) *industrial wastewater*, from industries using the same sewage system for their effluents (treated or not), and c) *rain-water*, particularly in the case of sewer systems constructed for both wastewater and storm-water (combined systems) (van Haandel and Lettinga, 1994). The sewage flow rate and composition vary considerably from place to place, depending basically on economic aspects, social behavior, type and number of industries located in the collection area, climatic conditions, water consumption, type and conditions of the sewer system, and so forth (Haskoning and Wageningen Agricultural University, 1994). Domestic wastewater is usually the main component of sewage, and is often used as a synonym. In this work, the term "raw sewage" will be used interchangeably with "sewage". When sewage is allowed to settle in a primary settler or other settling tank, the result is "settled sewage" or "pre-settled sewage".

Table 2 shows the most important constituents of raw sewage in three different cities (from van Haandel and Lettinga, 1994). Sand and coarse material (paper, dead animals, bottles, etc.) which are retained in the first steps of the treatment process (sand traps, screens) are not considered.

Provisions for the appropriate handling of sewage date as far back in time as the fourth century BC, judging by the "Athenian Constitution" written by Aristotle (van de Kraats, 1997). Thousands of years ahead, direct discharge to the environment is still the most common way of dealing with sewage and domestic wastewater, especially in developing countries. Yet several technological options are available today in the field of wastewater treatment, including conventional aerobic treatment in ponds, trickling filters and activated sludge plants (Metcalf and Eddy, 1991), direct anaerobic treatment (Lettinga, 1995; 1996), and resource-recovery wastewater treatments with biological systems, in which a combination of anaerobic and aerobic processes is applied (Jewell, 1996). Wastewater purification is the clearest paradigm of environmentally friendly technologies. Some negative aspects of development and urbanization can be diminished, or even eliminated, through a comprehensive treatment of domestic and industrial wastewater, directly and immediately enhancing the quality of the environment. Adequate wastewater treatment systems have to be simple in design and efficient in removing the pollutants. Energy consumption in these systems should be low, reuse of water and valuable by-products must be maximized, and the use of sophisticated equipment must be kept to a minimum. These features are required not only in the developing world, but also in industrial countries, where investment costs and energy consumption have to be reduced, while the treatment efficiency of the systems needs to be optimized.

Table 2. Composition of sewage in different cities (from van Haandel and Lettinga, 1994).

Constituent[a]	Pedregal (Brazil)	Cali (Colombia)	Bennekom (The Netherlands)
Settleable solids (ml/L)	8.2	---	---
Suspended solids			
Total	429	215	---
Fixed	177	106	---
Volatile	252	107	---
BOD	368	95	231
COD	727	267	520
Nitrogen (as N)	44	24	45
Organic	10	7	---
Ammonia	34	17	---
Phosphorus			
Total	11	1.3	18
Orthophosphate	8	---	14
Organic	3	---	4
Escherichia coli (number in 100 ml)	$4*10^7$	---	---
Sulfates	18	---	15
Chlorides	110	---	---
Alkalinity	388	120	350
Calcium	110	---	4
Magnesium	105	---	2
Temperature (°C)			
Maximum	26	27	20
Minimum	24	24	8

[a] Data in mg/L if not indicated otherwise. BOD = biological oxygen demand; COD = chemical oxygen demand.

ANAEROBIC SEWAGE TREATMENT

Anaerobic treatment of wastewater can be traced from the beginnings of wastewater treatment itself. Anaerobic processes have been used for the treatment of concentrated domestic and industrial wastewater for well over a century (McCarty, 1981; McCarty and Smith, 1986). The simplest, oldest, and most widely used process is the septic tank (Jewell, 1987). According to Buswell (1958), a tank designed to retain solids by means of sedimentation, similar to the septic tank, was firstly reported in 1857. Around 1860, a French engineer, Louis H. Mouras, built a closed chamber with a water seal in which all "excrementitious matter" was "rapidly transformed". This system, named "Mouras' Automatic Scavenger", was first described by Abbé Moigno in a report which appeared in France in 1881. This invention was enthusiastically defined at that time as "the most simple, the most beautiful, and perhaps, the grandest of modern inventions" (McCarty, 1985). A chronology of the development of anaerobic digestion for waste treatment can be found in Sastry and Vickineswary (1995). McCarty (2001) provided a summary of the development of anaerobic treatment, with some considerations about its future. The steep increase in energy prices in the 1970s reduced the attractiveness of aerobic methods, contributing to redirecting research efforts towards energy-saving alternatives like anaerobic treatment (van Haandel and Lettinga, 1994). At that time, discharge fees for industry were also introduced in some European countries (Mulder, 2003). McKinney (1983), Vochten *et al.* (1988), and Eckenfelder *et al.* (1988) have presented technical and economical comparisons between aerobic and anaerobic systems. A comparison among the most frequently used systems for wastewater treatment in developing countries, including stabilization ponds, activated sludge, trickling filters, anaerobic systems, and land disposal, was supplied by von Sperling (1996). Advantages and disadvantages of anaerobic sewage treatment, with special emphasis on high-rate reactors, are summarized in **Table 3** and **Table 4**, respectively.

High bacterial sensitivity to some environmental conditions (mainly pH, temperature, and toxic compounds), long starting processes, and the production of malodorous compounds, have been commonly cited as disadvantages of anaerobic treatment (Jewell, 1987). In fact, pH control may be needed for the treatment of some industrial wastewaters, but for other types of wastewater, including domestic wastewater and sewage, the composition is usually such that the pH will be kept in the optimum range without the need for chemical addition (van Haandel and Lettinga, 1994). Anaerobic bacteria can adapt quite easily to low temperatures, and high rate anaerobic treatment has been achieved at psychrophilic conditions (Kato, 1994, Kato *et al.*, 1994; Rebac *et al.*, 1995), including some experiences with sewage (Lettinga *et al.*, 1983b; Grin *et al.*, 1983; 1985; de Man *et al.*, 1986; 1988; Sanz and Fernández-Polanco, 1990; van der Last and Lettinga, 1992; Wang, 1994).

Table 3. Advantages of anaerobic wastewater treatment.

High efficiency. Good removal efficiency can be achieved in the system, even at high loading rates and low temperatures.
Simplicity. The construction and operation of these reactors is relatively simple.
Flexibility. Anaerobic treatment can easily be applied on either a very large or a very small scale.
Low space requirements. When high loading rates are accommodated, the area needed for the reactor is small.
Low energy consumption. As far as all plant operations can be done by gravity, the energy consumption of the reactor is almost negligible. Moreover, energy is produced during the process in the form of methane. The use of anaerobic systems can lead to a high degree of self-sufficiency.
Low sludge production. The sludge production is low, when compared to aerobic methods, due to the slow growth rates of anaerobic bacteria. The sludge is well stabilized for final disposal and has good dewatering characteristics. It can be preserved for long periods of time without a significant reduction of activity, allowing its use as inoculum for the start-up of new reactors.
Low nutrients and chemicals requirement. Especially in the case of sewage, an adequate and stable pH can be maintained without the addition of chemicals. Macronutrients (nitrogen and phosphorus) and micronutrients are also available in sewage, while toxic compounds are absent.
Suitable for campaign industries. Adapted anaerobic sludge can be preserved without feeding for a long time.

Table 4. Disadvantages of anaerobic wastewater treatment.

Low pathogen and nutrient removal. Pathogens are only partially removed, except helminth eggs, which are effectively captured in the sludge bed. The removal of nutrients is not complete and a post-treatment is sometimes required.
Long start-up. Due to the low growth rate of methanogenic organisms, the start-up takes longer as compared to aerobic processes, when no good inoculum is available.
Possible bad odors. Hydrogen sulfide is produced during the anaerobic process, especially when there are high concentrations of sulfate in the influent. A proper handling of the biogas is required to avoid bad smell. A significant proportion of the total amount of methane produced by the reactor may be dissolved in the effluent. Its recovery may be required to minimize smell nuisances and methane emissions to the atmosphere.
Necessity of post-treatment. Post-treatment of the anaerobic effluent is generally required to reach the discharge standards for organic matter, nutrients and pathogens.

On the other hand, anaerobic bacteria can tolerate a wide variety of toxicants (Speece, 1983). In fact, aerobic heterotrophs and methanogens showed similar sensitivities to toxicants, with the exception of an enhanced susceptibility of methanogens to chlorinated aliphatic hydrocarbons and chlorinated alcohols (Blum and Speece, 1991). Anaerobic sludge able to degrade pentachlorophenol (PCP), one of the biocides used in the United States to preserve wood products, was reported by Wu *et al.* (1993). Removals of PCP up to 99% were also reported by Hendriksen *et al.* (1992) using a glucose-supplemented continuous UASB reactor. UASB reactors have also been applied to rapidly detoxify and, under certain circumstances degrade, nitroaromatic compounds (Donlon *et al.*, 1996). The degradation of N-substituted aromatics, alkylphenols, and azo dyes under anaerobic conditions has also been demonstrated (Razo-Flores *et al.*, 1996; 1997; Razo-Flores, 1997; Donlon *et al.*, 1997).

The start-up of anaerobic reactors can be satisfactorily achieved in very short times if adequate inoculum is available (de Zeeuw, 1984), and this availability will be progressively greater, as anaerobic treatment plants are built and highly active anaerobic granular sludge becomes available for starting up new plants. Nonetheless, inoculation with active biomass was shown not to be a prerequisite to start-up anaerobic reactors for sewage treatment (Louwe Kooijmans and van Velsen, 1986), and many reactors started up without being inoculated at all, either at pilot scale (Schellinkhout *et al.*, 1985; Barbosa and Sant'Anna Jr., 1989), or full scale (Schellinkhout and Collazos, 1992; Draaijer, 1992). At low temperatures the start-up may take longer, but it can be successfully accomplished by inoculating the reactor with digested sludge (Singh *et al.*, 1997).

Finally, an adequate construction of the reactor and a proper operation can eliminate completely the problem of bad odors in anaerobic reactors (Conil, 1996).

As we can see, substantial improvements have been made in tackling most of the alleged disadvantages of anaerobic treatment, with the result that only a few of the previously presumed drawbacks have remained, while all its principle benefits over conventional aerobic methods still apply (Lettinga *et al.*, 1987; Lettinga, 1995; Lettinga, 1996; 1996b; 2001).

According to Jewell (1985), "*there is little doubt that development of a cost-effective and efficient anaerobic sewage treatment alternative would be one of the most significant advances in waste treatment history*". Lettinga *et al.* (1987) fully agreed with this statement by saying that *"...a satisfactory application to raw domestic sewage would represent the maximum possible accomplishment for high-rate anaerobic treatment systems"*. The term "high-rate" was once used for the later designs of sewage sludge digesters, but it is now widely used to refer to anaerobic treatment systems meeting at least the following two conditions: a) high retention of viable sludge under high loading conditions, and b) proper contact between incoming wastewater and retained sludge (Lettinga *et al.*, 1987). Anaerobic treatment in high-rate reactors is increasingly recognized as the core method of an advanced technology for environmental protection and resource preservation, and it represents, combined with other proper methods, a sustainable and appropriate wastewater treatment system for developing countries (Lettinga *et al.*, 1987; 1993; van Buuren, 1996; Lettinga, 1996; 1996b; Lettinga *et al.*, 1997; Verstraete and Vandevivere, 1999). It is often questioned why aerobic treatment of sewage is not replaced more rapidly by the economically more attractive and the conceptually more holistic direct anaerobic treatment (Mergaert *et al.*, 1992). Anaerobic treatment would provide tremendous advantages over conventional aerobic methods. The costs of aeration and sludge handling, the two largest costs associated with aerobic sewage treatment, would be reduced dramatically because a) no oxygen is needed in the process and b) the production of sludge is 3 to 20 times smaller than in aerobic treatment (Rittmann and Baskin, 1985). Moreover, the sludge (biomass) produced in aerobic processes has to be stabilized in classic anaerobic sludge digesters before it can be safely disposed of, even though it was shown to be resistant to anaerobic degradation (Sanders *et al.*, 1996).

Some characteristics of sewage, like low COD concentration, high fraction of COD as SS, relatively low temperature, and load fluctuations, are particularly relevant to anaerobic treatment and can have a negative impact on the process performance or costs, exaggerating the difficulty of treatment by anaerobic processes (Jewell, 1987). The impact of some of these characteristics was studied by Rittmann and Baskin (1985), who proposed modeling approaches for the purpose of making quantitative evaluations. Careful selection of the technology and appropriate reactor design and operation have overcome most of these possible difficulties.

THE UASB REACTOR

In spite of their early introduction, the interest on anaerobic systems as the main biological step (secondary treatment) in wastewater treatment was scarce until the development of the UASB reactor (**Figure 2**, left).

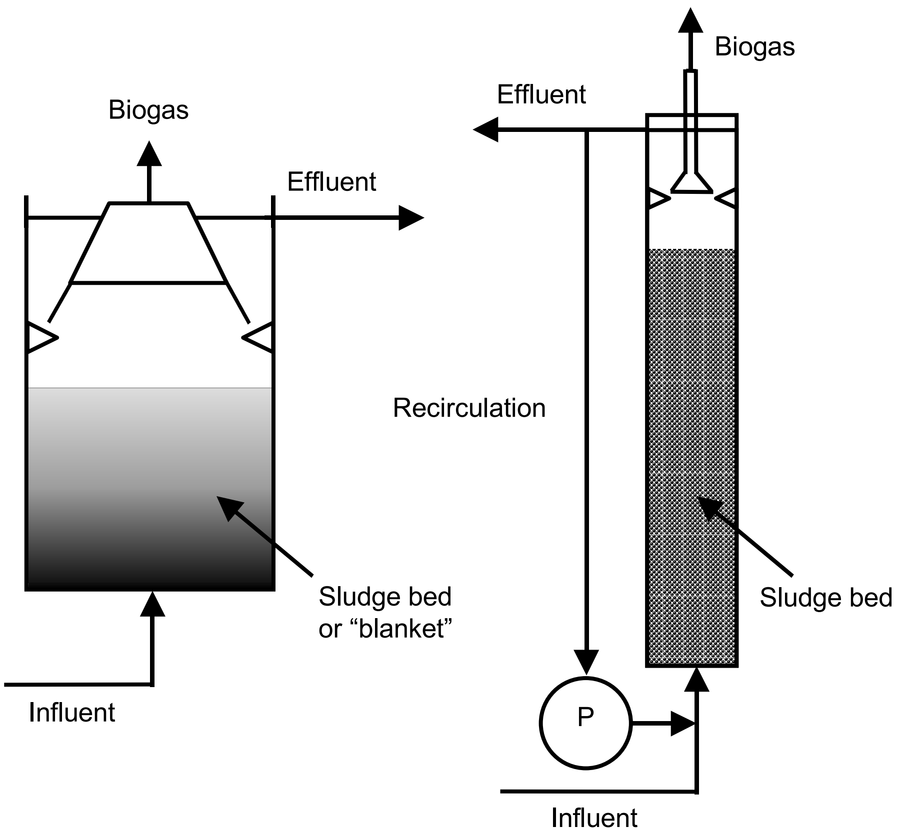

Figure 2. Schematic diagrams of UASB (left) and EGSB (right) reactors. Modified from van Haandel and Lettinga (1994) and Wang (1994). P = recirculation pump.

Antecedents of the UASB reactor can be found in the so-called anaerobic contact process (Coulter *et al.*, 1957; Ettinger *et al.*, 1958; Fall and Kraus, 1961; Simpson, 1971; Pretorius, 1971), the Imhoff tank, an improved kind of septic tank, and the reversed flow Dorr-Oliver Clarigester used in South Africa in the fifties and sixties (Lettinga, 2001). Winslow and Phelps (1911) had previously used a similar system called the "biolytic tank" in 1910. Now, the UASB reactor is extensively used for the treatment of several types of wastewater (Hulshoff Pol and Lettinga, 1986; Lettinga and Hulshoff Pol, 1991; Kato *et al.*, 1994; Lettinga 1995; 1996; 1996b). The success of the UASB concept relies on the establishment of a dense sludge bed in the bottom of the reactor, in which all biological processes take place. This sludge bed is basically formed by accumulation of incoming suspended solids and bacterial growth. In upflow anaerobic systems, and under certain conditions, it was also observed that bacteria could naturally aggregate in flocs and granules (Hulshoff Pol *et al.*, 1983; Hulshoff Pol, 1989). These dense aggregates have good settling properties and are not susceptible to washout from the system under practical reactor conditions. Retention of active sludge, either granular or flocculent, within the UASB reactor enables good treatment performance at high organic loading rates. Natural turbulence caused by the influent flow and the biogas production provides good wastewater-biomass contact in UASB systems (Heertjes and van der Meer, 1978). Higher organic loads can be applied in UASB systems than in aerobic processes (Kato, 1994). Therefore, less reactor volume and space is required while, at the same time, high-grade energy is produced as biogas. Several configurations can be imagined for a wastewater treatment plant including a UASB reactor. In any case, there must be a sand trap, screens for coarse material, and drying beds for the sludge. The UASB reactor may replace the primary settler, the anaerobic sludge digester, the aerobic step (activated sludge, trickling filter, etc.), and the secondary settler of a conventional aerobic treatment plant. However, the effluent from UASB reactors usually needs further treatment, in order to remove remnant organic matter, nutrients and pathogens. This post-treatment can be accomplished in conventional aerobic systems like waste stabilization ponds (WSP). The economics of anaerobic treatment in UASB reactors were thoroughly discussed by Lettinga *et al.* (1983).

THE EGSB REACTOR

Tracer studies demonstrated that internal mixing was not optimal in a pilot-scale UASB reactor treating sewage at temperatures ranging from 4 to 20°C (de Man *et al.*, 1986). This produced dead space in the reactor, leading to a reduction in the treatment efficiency. In order to improve the sludge-wastewater contact and to use the entire reactor volume efficiently, a better influent distribution was required. Different feed inlet devices, more feed inlet points per square meter or higher superficial velocities have been proposed as solutions. The use of effluent recirculation combined with taller reactors (or a high height/diameter ratio), resulted in the expanded granular sludge bed (EGSB) reactor (**Figure 2**, right), where a high superficial velocity is applied (van der Last and Lettinga, 1992). The main characteristics of EGSB reactors are presented in **Table 5**.

In this reactor concept, the liquid upflow velocity (V_{up}) (> 4 m/h) causes the granular sludge bed to expand, eliminating dead zones and resulting in better sludge-wastewater contact. However, a direct relationship between upflow velocity and substrate consumption could not be found, and the granule size and inner structure seem to play a more relevant role in fully expanded EGSB reactors (González-Gil *et al.*, 2001). Accumulation of flocculent excess sludge between the sludge granules is also prevented (van der Last and Lettinga, 1992). Soluble pollutants are efficiently treated in EGSB reactors but suspended solids are not substantially removed from the wastewater stream due to the high upflow velocities applied. Recirculation of the effluent dilutes the influent concentration, but it was extensively proven that low strength wastewater can efficiently be treated in EGSB reactors (Kato *et al.*, 1994; Kato, 1994). Influent dilution may also allow the treatment of toxic compounds in these reactors. In UASB reactors, the sludge bed behaves more or less as a static bed, but in fully expanded EGSB reactors, it is considered as a completely mixed tank (Rinzema, 1988). Compared to UASB reactors, higher organic loading rates (as $kgCOD/m^3_{reactor}.d$) can be accommodated in EGSB systems. Consequently, the gas production is also higher, improving even more the mixing process inside the reactor. The exact mixing pattern cannot be generalized, and it must be evaluated in each reactor by assessing the reactor hydrodynamics (van der Meer, 1979). In tall reactors, the gas loading (in $m^3/m^2.h$) and the hydrostatic pressure at the bottom can be higher than in short reactors and the effect of these parameters on the performance of the process has to be considered as well.

Table 5. Characteristics of EGSB reactors.

- Higher upflow velocities (in the range of 4 to 10 m/h), and organic loading rates (up to 40 $kgCOD/m^3.d$) are applied, compared to UASB reactors.
- The sludge bed is expanded.
- More suitable for dilute wastewater than UASB reactors (in that case effluent recirculation is not applied).
- The sludge is always granular, very active, and the settleability is good.
- The mixing pattern is different from UASB reactors, due to the higher V_{up} and the increased gas production (m^3 of gas/m^2 of reactor area), leading to a different sludge-wastewater contact.
- The hydrostatic pressure on the sludge at the bottom may be greater if the reactor is tall, but its effect on reactor efficiency and biomass growth is not well understood yet.
- Flocculent sludge is washed-out of the reactor.
- No good removal of suspended solids and colloidal matter can be achieved.

EXAMPLES OF SEWAGE TREATMENT IN UPFLOW REACTORS

Low temperatures

There are clear indications that UASB reactors can cope with sewage temperatures around 18ºC and lower for prolonged periods without a substantial reduction in their treatment efficiency (Haskoning, 1996; 1996b). The application of UASB reactors to sewage treatment under low temperature conditions has been studied in The Netherlands since 1976 (Lettinga *et al.*, 1981; Grin *et al.*, 1983; 1985; de Man *et al.*, 1986; van Velsen and Wildschut, 1988). Lettinga *et al.* (1983b) reported results obtained in laboratory UASB reactors with raw domestic sewage using digested sewage and sugar beet cultivated sludge as seed material. Some of their results are summarized in **Table 6**. At the same time, a 6 m^3 UASB reactor seeded with digested sewage sludge was operated at HRT of 14-17 h. COD reduction reached 85-65% and 70-55% at 20 and 13-17°C, respectively. They concluded that the UASB concept was a simple, compact, and inexpensive technology for sewage treatment, even at relatively low temperatures. Fernandes *et al.* (1985) confirmed their results using two small (12.4 L capacity) UASB reactors to treat settled domestic sewage. Based on research carried out in The Netherlands on different UASB reactors (0.120, 0.240, 6, and 20 m^3), de Man *et al.* (1986) concluded that anaerobic treatment of raw domestic sewage (COD = 500-700 mg/L) can be accomplished at 12-18°C applying HRTs of 7-12 h with total COD and BOD removal efficiencies of 40-60%, and 50-70%, respectively. However, at that time this performance was not considered attractive to treat sewage under Dutch conditions. Sludge-wastewater contact was found to affect considerably the treatment efficiency, especially at low temperatures when the gas mixing was poor. Better SS removals were achieved using shallow reactors. More details are provided in **Table 6**. Nonetheless, efficiencies up to 80% were obtained at 10-20°C when treating sewage from a separated sewer system (domestic wastewater separated from rainwater) with a granular bed reactor (de Man *et al.*, 1988). Higher upflow velocities, like those applied in EGSB reactors, induce a better sludge-wastewater contact and the removal efficiency of soluble substrates is likely to increase (de Man *et al.*, 1988). Van der Last and Lettinga (1992) described experiments performed in Bennekom (The Netherlands) using 120 and 205 L EGSB reactors treating settled sewage at ambient temperatures (15-20°C in summer and 6-9°C in winter), with HRTs ranging from 2 to 7 hours. Influent total COD and paper-filtered COD were about 400 and 300 mg/L, respectively. Different removal efficiencies were observed under dry and rainy weather conditions, but never exceeding 50% of the soluble COD fraction. The COD removal rates (related to soluble COD) deteriorated during winter conditions and this fact was attributed to either low sludge methanogenic activity at low temperatures, or a limitation in the maximum possible acidification of the soluble COD fraction in the presettled sludge. VFA (volatile fatty acids) measurements, which could have helped to decide the cause of this decrease in removal efficiency, were not reported. Poor removal of SS was observed in EGSB reactors, due to the high upflow velocities applied (4-8 m/h).

A combination of a 47-L UASB reactor and an aerobic post-treatment named the "hanging sponge cubes" was evaluated by Agrawal *et al.* (1997) for the treatment of raw sewage (total COD = 300 mg/L) in a climate with ambient temperatures ranging from 7 to 30°C. During winter time the raw sewage temperature was 2 to 4°C higher than the ambient temperature. The UASB reactor was seeded with digested sewage sludge and operated for more than 2 y at an HRT of 7 h. Total COD removal was about 70% throughout the year. The hanging sponge cubes posttreatment process provided biological removal of organics and nitrogen even at high loading rates. Further research is recommended on several aspects of the hanging sponge cubes posttreatment process.

Development of granules was observed in a UASB reactor treating domestic sewage at 20ºC at HRTs from 40 h during start up to 10 h at the end of a 9-month period (Singh and Viraraghavan, 1998) (results in **Table 6**). The performance of a UASB reactor for sewage treatment under moderate to low temperature conditions (25 to 13ºC) was assessed by Uemura and Harada (2000) (**Table 6**). A high total COD removal efficiency was achieved (around 70%), irrespective of the operational temperature, although hydrolysis of the solids retained in the sludge bed was significantly affected by temperature. Elmitwalli (2000) investigated the treatment of raw and settled sewage at 13°C in a UASB reactor and a so-called anaerobic hybrid (AH) reactor. The AH reactor used was basically an upflow reactor in which a synthetic filter medium replaced the gas-solid-liquid separator, typical of UASB reactors, at the upper part. The medium consisted of vertically oriented reticulated polyurethane foam sheets with knobs at one side. Removal efficiencies obtained with both types of reactors were relatively high (results for the UASB reactor are shown in **Table 6**). However, the treatment of raw sewage in the UASB reactor seemed to be not practical due to sludge bed flotation (attributed to the high concentration of suspended COD in the influent). The presence of reticulated polyurethane foam sheets in the AH reactor prevented sludge bed flotation. The removal of colloidal and dissolved COD was significantly higher when the reactors were fed with settled sewage. No sludge flotation was observed when the UASB reactor treated settled sewage. Mahmoud (2002) studied the application of UASB reactors for sewage treatment at a sewage temperature of 15ºC, the average sewage temperature in the Middle East countries during wintertime (**Table 6**). The performance of a single-stage UASB reactor was improved by digesting the excess sludge in an anaerobic digester at 35ºC, and recirculating the sludge back into the reactor. The performance of the system UASB-Digester was as good as that achieved in tropical countries with single-stage UASB reactors, and the wasted sludge was much more stabilized. Promising results were also obtained in Jordan with UASB reactors treating strong raw sewage at a temperature of 18ºC in winter and 25ºC in summer (Halalsheh, 2002). A comparison was made between one and two-stage systems. **Table 6** shows average results obtained during winter time with the first stage of the two-stage system, and the one-stage reactor. A higher degree of sludge stabilization was observed in the one-stage reactor, compared to the first stage of the two-stage system.

High temperatures

The full-scale application of UASB reactors to domestic wastewater has been a success in tropical areas, where mean sewage temperature can go up to 30ºC (Alaerts *et al.*, 1990; Elmitwalli, 2000; Mahmoud, 2002; Foresti, 2002). A joint project financed by the Dutch government was carried out in Cali (Colombia) to test the technical and financial feasibility of the UASB process for sewage treatment at pilot scale and to develop design criteria that could be transferred to Colombian institutions in order to further promote the use of this technology in developing countries (Schellinkhout *et al.*, 1985; Louwe Kooijmans and van Velsen, 1986). The plant built in Cali is claimed to be the first of its kind in the world (Louwe Kooijmans *et al.*, 1985). A 64 m^3 reactor was operated at an average sewage temperature of 25°C. Diluted digested cow manure was used as inoculum, and the plant was fully operational after 6 months at an HRT of 8 hours. It was concluded that, under Cali conditions (rather septic sewage), HRTs of 4-6 hours with only 1 inlet point every 4 m^2 give satisfactory results. Sewage in Cali was rather dilute (267 mgCOD/L, 95 mgBOD/L) when compared to domestic sewage in European countries, while the COD/BOD ratio of the influent was higher (2.1-4.4) due to the presence of high concentrations of total and volatile SS (215 mgTSS/L, 108 mgVSS/L) (Lettinga *et al.*, 1987). COD and BOD removal efficiencies higher than 75% were observed while SS removal was about 70%. The UASB process was found to be economically more attractive than facultative ponds and oxidation ditches (the "carrousel" system), especially when capital costs were included. A comprehensive assessment of the feasibility of anaerobic sewage treatment in sanitation strategies in developing countries was presented by Alaerts *et al.* (1990; 1993).

Barbosa and Sant'Anna Jr. (1989) reported results from 9 months of operation of a 120-L UASB reactor treating raw sewage with 627 mgCOD/L and 357 mgBOD/L, at ambient temperatures (19-28°C). The startup was successfully achieved without inoculation and the reactor was operated at an HRT of only 4 h throughout the entire experimental period. Spherical bacterial granules were observed after one month of operation. Granules up to 8 mm in diameter were observed at the end of the experiment. COD, BOD and TSS removal increased steadily during the first 4 months of operation. After that, the removal efficiency improved more slowly, indicating the end of the startup period. The fast evolution of the sludge bed was attributed to the high content of suspended organic matter in the incoming sewage (76% of the total influent COD). After the startup phase was over (last 5 months of operation), total BOD removal of around 78% was achieved, while total COD removal reached 74%. The reported accumulation of solids in the sludge bed was almost 50% faster in the last 5 months of operation than during the startup. Interestingly, SS concentration in the effluent did not depend on the variations observed in the influent, an observation also reported by Wang (1994). The average SS removal during this period was 72%. COD was mainly removed through physical processes of SS retention in the sludge bed. Dissolved COD was responsible for most of the methane production, but from COD balances it was apparent that some methane was also produced from the hydrolysis and fermentation of entrapped particulate organic matter. The high content of slowly degradable undissolved organic matter of the sewage used in this experiment and the high capacity of solids retention in the UASB reactor, promoted an excess sludge production estimated to be 18 kg PE^{-1} $year^{-1}$ (PE = population equivalent; one PE is the mean amount of oxygen binding compounds discharged with the wastewater per inhabitant per day) which has to be removed from the reactor and further treated, dried, or disposed of.

Other studies showed good COD removal efficiency using a glucose-enriched domestic wastewater. However, in the latter case, self-inoculation was not achieved at a temperature of 30°C (Gnanadipathy and Polprasert, 1993). The reactors only performed satisfactorily when inoculated, particularly with facultative WSP sludge or anaerobically digested sludge. A "partitioned" UASB reactor was assessed by Chernicharo and Cardoso (1999) for the treatment of domestic sewage from small villages in Brazil (**Table 6**). This system was constituted of three digestion compartments, three gas separation devices, and a single settler compartment. The distribution of the (variable) incoming flow rate into one, two, or three digestion chambers allowed the establishment of more stable upflow velocities and less occurrence of dead zones. Apparently, a better contact between substrate and biomass was achieved and an improved performance was observed, compared to conventional one-chamber UASB reactors working under similar conditions. Cavalcanti (2003) presented results from the operation of a typical UASB reactor and another with a different phase separator design, consisting of a series of parallel plates above the conventional separator (results from the normal UASB reactor were summarized in **Table 6**). The proposed separator design doubled the treatment capacity of the reactor.

Full-scale plants

Full-scale application of the UASB process has been successfully implemented in several countries. Bilateral co-operation between India and The Netherlands led in 1985 to the design and construction of a full-scale UASB reactor for domestic sewage in the town of Kanpur (India) which has been in operation since April 1989. This plant was designed to treat 5000 m^3 of raw sewage per day. Results obtained during a monitoring period of 12 months were reported by Draaijer *et al.* (1992). The startup was carried out without inoculum. After sufficient sludge accumulated in the reactor, the sludge quality was reported to be improved by stopping the feeding of the plant for approximately 2 weeks. The period of startup was about 10 weeks. COD, BOD and TSS removals of respectively 74, 75, and 75% were achieved at a nominal HRT of 6 h. However, in order to meet Indian standards for discharge on surface waters (30 mgBOD/L; 50 mgTSS/L) posttreatment was required. Sludge washout was prevented to a large extent by the presence of baffles in the effluent overflow gutters, especially when sludge concentrations in the reactor were "very high" (sic; no values given). Results further showed that one influent inlet point every 3.7 m^2 sufficed to ensure good distribution of the wastewater over the bottom of the reactor. Low temperatures did not affect treatment efficiencies in winter. However, biogas production decreased in the coldest period (3 weeks), when the temperature reached 20°C, but recovered when temperature increased again. SS concentration was higher at the bottom of the sludge bed, suggesting that the amount of solids retention could have been somewhat increased by discharging excess sludge from high levels. The sludge drying characteristics were good, and excess sludge could be dried within 6 days up to 300-500 gTSS/kg of sludge. Large fluctuations in the influent COD, BOD, and TSS concentration were attributed to the presence of a cluster of 150 tanneries in a nearby area, operating on a seasonal basis. Tannery wastewater also contributed considerably to the sulfate load of the system, and sulfide was regularly observed in the effluent. Another treatment plant was built in Kanpur to treat the wastewater of approximately 180 tanneries after dilution with domestic wastewater in a ratio 1:3 (Haskoning, 1996). The design flow of this plant was 36000 m^3/d. The plant was started in April 1994 and the startup period lasted approximately 5 months. The reactor temperature varied from 18°C in winter to 32°C in summer. Results of more than one year of operation are presented in **Table 6**. Sulfate reduction was incomplete, and sulfate concentrations up to 200 mg/L were detected in the effluent. The average loading rate during the period was 2.5 kgCOD/m^3.d. COD, BOD, and TSS removals were respectively 50-70%, 50-65%, and 45-60%. VFA concentration in the effluent was low. Removal of SS from the effluent is the main step to be taken for a further optimization of the plant efficiency.

Based on the results obtained in Kanpur, a full-scale UASB plant with a posttreatment facility consisting of a pond with an HRT of one day was designed for the Indian town of Mirzapur. This plant has been constructed as part of the Indo-Dutch Environmental and Sanitary Engineering Project under the Ganga Action Plan, and has been in full operation since April 1994 (Haskoning, 1996b). Operation results in the period April 1995 to March 1996 are presented in **Table 6**. The reactor temperature also varied from 18°C in winter to 32°C in summer, as in the case of Kanpur described above. The average loading rate during the reported period was about 0.95 kgCOD/m^3.d for the UASB reactors and 0.13 kgCOD/m^3.d for the polishing pond. In this pond the remaining SS were retained, in such a way that the final effluent complied with the Ganga action plan standards (BOD = 30 mg/L, TSS = 50 mg/L). The overall removal efficiency of the Mirzapur wastewater treatment plant for COD, BOD and TSS was about 81, 86 and 89%, respectively (Haskoning, 1996b). A similar plant designed for 50000 m^3/d was built in the Indian city of Hyderabad (Tare *et al.*, 1997). In October 30, 1990, a huge sewage treatment plant based on UASB technology was started up in Bucaramanga, Colombia (Schellinkhout *et al.*, 1988; Schellinkhout and Collazos, 1992). This plant was designed to treat 31000 m^3/d and was the largest of its kind at that time. The plant consisted of two UASB reactors operating in parallel and a facultative pond in series as posttreatment. Based on 3 y operation, COD removal efficiency ranging from 70 to 77% was reached, the UASB reactors being responsible for 45 to 50% of that removal (Schellinkhout and Osorio, 1994). No inoculum was used and the startup period lasted around 6 months. In **Table 6**, a detailed description of operational characteristics is given. Profuse details about design, construction, operation, maintenance, and follow-up of the plant can be found in Collazos Chávez (1991), Hoyos Carrillo (1991), and Cala (1991). The design of this full-scale treatment plant was based on 4 y operation of a 35 m^3 pilot-plant UASB reactor (Schellinkhout and Collazos, 1992) (see also **Table 6**). Based on the results obtained in Bucaramanga, a sewage treatment plant consisting of a combination of UASB reactors and trickling filters or ponds as posttreatment was suggested as an attractive solution for Egypt, where the sewage temperature is lower but the concentration is higher than in Colombia (Schellinkhout, 1993). After conducting batch digestibility assays the use of anaerobic digestion as a pretreatment for sewage was also considered feasible for Egypt by Bellamy *et al.* (1988).

In Brazil, after promising results were obtained at pilot-scale (106 L), a 120 m^3 UASB reactor for domestic sewage treatment was designed and constructed (Vieira and Souza, 1986; Vieira, 1988). This reactor operated for 5 y at ambient temperature. Results of two years of tests are presented in **Table 6** (Vieira and Garcia Jr., 1992). Interestingly, round or ellipsoidal sludge granules up to 2 mm were observed during all the studied periods. Based on the experience gathered with this reactor, several design recommendations were made (Souza, 1986). In 1987, a 336 m^3 demonstration plant was built in Senigallia (Italy) (Urbini *et al.*, 1988; Collivignarelli *et al.*, 1991) (results in **Table 6**). This plant treated sewage at temperatures ranging from 7 to 27°C from a strongly fluctuating population (from 2000 inhabitants in winter to 20000 in summer) (Maaskant *et al.*, 1991). In Odemira, southern Portugal, Portuguese-Dutch co-operation led to the construction of a 20-m^3 UASB demonstration plant (Maaskant *et al.*, 1991). The planned average HRT was set at 10 h. This plant was designed to handle the entire sewage flow of a community of 320 inhabitants, with plans to extend its capacity to cope with the sewage of nearby tourist areas. Florencio *et al.* (2001) described the performance of a UASB reactor for sewage treatment in Recife, Brazil, during a considerable period of time. This reactor performed well regardless of operating problems and influent fluctuations (**Table 6**). It was concluded that regular maintenance, cleansing, removal of grit from raw sewage, and periodic withdrawal of the scum layer from the reactor were important operational parameters in order to guarantee a safe and efficient operation. A thorough review of the state of the anaerobic technology for wastewater treatment in Mexico was presented by Monroy *et al.* (2000). About 40% of all anaerobic reactors built in this country (85 in total) are currently treating domestic wastewater, including the biggest UASB reactor ever built (83700 m^3, to be extended to 133920 m^3 in the future), which treats a combination of industrial and municipal wastewater. However, most of the full-scale UASB reactors treating sewage in Mexico are small, with only 3 plants bigger than 350 m^3.

The study and application of UASB technology to domestic sewage is rapidly growing in Latin America, with reports in Brazil (de Sousa and Foresti, 1996; Chernicharo and Borges, 1997) (see **Table 6**), Colombia (Maaskant *et al.*, 1991; Mora B. and Sterling S., 1996; Rodríguez *et al.*, 2001), Guatemala (Conil *et al.*, 1996), and Mexico (Monroy *et al.*, 2000). Encouraging results have also been obtained in the Mediterranean area (Urbini *et al.*, 1988; Collivignarelli *et al.*, 1991). A project financed by the European Union studied the application of UASB reactors to sewage treatment in several locations, including The Netherlands, Spain, Jordan, Egypt, and Hebron (García Encina *et al.*, 1996; Berends, 1996). A comprehensive data bank on anaerobic sewage treatment plants was compiled by the German development co-operation agency (GTZ), with emphasis on Latin America and South East Asia (Hulshoff Pol *et al.*, 1997). However, although substantial experience on the design and operation of UASB reactors for sewage treatment has been gathered lately, most of the results have not yet been published (Wiegant, 2001). All in all, the UASB reactor appears as the most robust of all the anaerobic treatment processes, and “is by far the most widely used high-rate anaerobic system for sewage treatment” (van Haandel and Lettinga, 1994).

On-site treatment

Application of modified UASB reactors for single households in isolated locations, like farms and recreational facilities not connected to the centralized sewerage system, was studied under Dutch (low) and Indonesian (high) ambient temperatures. In The Netherlands, Bogte *et al.* (1988; 1993) tested three 1.2-m^3 UASB reactors in different rural locations with varying results (**Table 6**). A similar configuration was tested in Bandung (Indonesia) by Lettinga *et al.* (1993). Treatment efficiencies in Indonesia were in general very high (see **Table 6**), while good sludge stabilization and high sludge hold-up were achieved. These systems were called UASB-septic-tanks, because they shared features of both methods. Sludge gradually accumulates in the reactor, as in septic tanks, but they are operated in upflow mode, as UASB reactors. The design is almost as simple as that of conventional septic tanks but the treatment efficiency is much higher (Zeeman, 1997). These reactors should startup in summer with an inoculum of at least 15% of the volume, according to the results of Zeeman (1991), that worked on manure digestion at low temperatures in accumulation systems. Suggestions for improving the treatment efficiency in these systems include the use of two- or three-stage UASB reactors and the adoption of proper posttreatment methods, like small aerobic lagoons (Lettinga *et al.*, 1993). The UASB process was also applied to treat sewage from small-size communities (235 houses) in Brazil (Vieira *et al.*, 1994). Being a low-income neighborhood, sewage concentration was relatively high, with 402 mgCOD/L, 515 mgBOD/L (sic; BOD can never be higher than COD) and 379 mgTSS/L. Average removal efficiencies of 74, 80, and 87% were obtained for COD, BOD, and TSS respectively. Granulation was observed after 6 months of operation. The feasibility of an integrated waste management system for new urban areas which combines anaerobic digestion in conventional stirred-tank reactors (receiving black water from vacuum toilets, and shredded kitchen and garden biowaste) and aerobic biofilm systems like aerated sand-filters, rotating disks plants and trickling filters (for graywater treatment) was tested in Lübeck, Germany (Otterpohl *et al.*, 1997). This system could replace the centralized sewerage system for a settlement of about 300 inhabitants. UASB-type reactors were not planned in this particular study, but could well be included in decentralized or semi-centralized systems like this. A pilot-scale single-step community on-site UASB reactor was operated for a long period in a University in Tanzania (Mgana, 2003). Results showed that it is feasible to attain high COD removal efficiencies (the average was 64%) provided that the problem associated with sludge flotation is properly addressed (more details in **Table 6**). When a second UASB reactor was connected in series, COD removal efficiencies close to 70% were achieved in the system, at an overall HRT of 7.4 h (5 + 2.4).

Two-step processes

Two-stage anaerobic processes have been proposed to retain and degrade suspended solids from sewage at lower temperatures, like those prevailing in moderate climates (van Haandel and Lettinga, 1994; Wang, 1994). In the first stage, the particulate organic matter is entrapped and partially hydrolyzed into soluble compounds, which are then digested in the second stage. The removal efficiency of suspended solids in the first reactor will be higher than that of organic matter and excess sludge needs to be discharged regularly. As a result of that, the sludge age remains relatively low in this reactor, hindering the development of the slow-growing methanogens and reducing methanogenesis to a minimum. Moreover, the development of acid fermentation may tend to depress the pH to a value below the optimum range for methanogenic bacteria. In the effluent of the first reactor the organic matter will be present predominantly as dissolved compounds. Accumulation of biodegradable solids in the first reactor may occur at low temperatures, when the hydrolysis rate becomes limiting. The excess sludge of such a reactor can be further stabilized in a separate, heated sludge digester (van Haandel and Lettinga, 1994; Wang, 1994; Mahmoud, 2002). The stabilized sludge can be separated in a liquid-solid separation step and the liquid phase, enriched with soluble organic matter, can be mixed with the effluent of the unheated hydrolytic reactor and be submitted to methanogenic treatment in a second UASB or EGSB reactor. Part of the sludge can be returned to the anaerobic reactor in order to improve its methanogenic capacity.

A process consisting of a sequential HUSB reactor followed by an EGSB reactor, combined with an additional sludge stabilization tank was presented by Wang (1994). The latter tank, named "sludge recuperation tank" was a semi-continuous anaerobic digester, gently stirred at 60 rpm, operating at different temperatures with an HRT of 2 days. The total process provided 71% COD and 83% SS removal efficiencies at temperatures above 15°C, and 51% COD and 77% SS removal at 12°C. Over 50% hydrolysis of the removed SS was obtained in the HUSB reactor at higher ambient temperatures (exceeding 19°C). HRTs applied were 3 and 2 h for the HUSB and EGSB reactors respectively, and two days for the sludge recuperation tank. The EGSB reactor removed up to 32-60% of the soluble COD at 9-21°C. This concept looks attractive for sewage treatment, especially due to the short HRTs needed for COD and SS removal and sludge stabilization. The HUSB reactor can be considered as a relatively highly loaded UASB system for the removal and hydrolysis of suspended COD. The HRT in the HUSB reactor is very similar to that applied in primary sedimentation tanks, but the removal efficiencies of COD, BOD and SS are considerably higher (Wang, 1994). A mathematical model representing the HUSB reactor was proposed by Rijsdijk (1995) and additional work on the solids removal efficiency and hydrolysis ratio in HUSB reactors has been carried out by Berends (1996) and Sanders *et al.* (1996). A similar configuration, called UASR (upflow anaerobic solids removal) reactor, coupled to an upflow sludge digester was investigated by Corstanje (1996) for the treatment of waste activated sludge under anaerobic conditions. In UASR reactors only SS removal is obtained, as in normal settling tanks, while in HUSB reactors hydrolysis also takes place. Therefore, more sludge has to be discharged in principle from UASR than from HUSB reactors. Removal and pre-hydrolysis of suspended COD from raw sewage, waste activated sludge, and dairy wastewater in UASR reactors was reported by Zeeman *et al.* (1996). At 17°C, with an HRT of 3 h, 65% of the suspended COD was removed from raw sewage. The retained sludge was only partially hydrolyzed, and it should undergo further digestion at higher temperature.

Sayed and Fergala (1995) and Fergala (1995) also studied the feasibility of a two-stage anaerobic system for sewage treatment. The first stage consisted of two flocculent UASB reactors operated intermittently while the second stage was a UASB reactor seeded with granular sludge. The first stage was intended to remove and partially hydrolyze SS and the second was devoted to the removal of soluble organic material. It was claimed that intermittent operation of the first stage provides further stabilization of the removed solids. The experiments were carried out at an ambient temperature of 18-20°C and average HRTs of 8-16 h for the first stage (taking both reactors into account) and 2 h for the second stage. COD and BOD removal efficiencies up to 80 and 90%, respectively, were achieved. Most of the removal took place in the first stage.

Two-stage UASB reactors were also applied to domestic sewage in Spain (García Encina *et al.*, 1996) with overall COD reductions up to 62% at 14 h of HRT, working at temperatures ranging between 9 and 26°C. A pilot-scale combination of an UASB reactor and a packed bed proved to be efficient in removing total COD and SS from raw sewage in Puerto Rico (Tang *et al.*, 1995). Average total COD, BOD, and SS removals of around 80, 87, and 95% were achieved with the entire system, applying HRTs ranging from 6 to 24 h. From the total removal of organic pollutants, more than 70% COD and 80% SS were removed by the UASB column. Additional data are provided in **Table 6**. The working temperature was not reported, but the sewage was identified as "warm and dilute". Some of the removed COD accumulated in the UASB columns and was not immediately converted to methane. Periodic removal of 50% of the sludge resulted in an increase in the gas production rate. According to this study, in a full-scale plant, withdrawal of sludge could be done once every 3 to 6 months from the UASB and once every 1 or 2 y from the packed bed. A gravel filter was also tested as a final polishing step, but it was shown not to be relevant in the overall treatment efficiency. Some gas production dips were observed during the experiments, but the cause could not be identified properly. Sometimes, brewery wastewater in the influent also affected the gas production. The COD removed was in general higher than the gas produced. This was attributed to retention of undegraded SS in the sludge bed, but dissolved methane in the effluent could have contributed as well.

Kalogo and Verstraete (1999; 2000) proposed a system that includes chemically enhanced primary sedimentation to remove suspended solids from raw sewage, particularly for tropical areas. The supernatant is treated in a UASB reactor, while the primary sludge is digested separately in a conventional sludge digester. The effluent from the UASB reactor is disinfected with ozone (which could be produced with electricity generated from the biogas coming out of the sludge digester), and eventually submitted to secondary sedimentation prior to its potential (re)use in agriculture. Elmitwalli (2000) studied a two-step system consisting of an anaerobic filter (AF) plus an AH reactor (a UASB reactor with a filter on top) at a sewage temperature of 13ºC. Total COD removal efficiency was as high as 71%, similar to values found in tropical areas. Removal of suspended and dissolved COD was highest when HRTs of 4 h were applied in both steps. Based on these results, a two-step system consisting of a highly loaded UASB reactor and a hybrid system was proposed as an attractive treatment method for raw sewage treatment at low temperatures (around 13°C), especially for community on-site applications. Halalsheh (2002) studied different two-stage configurations in Jordan.

A system UASB + UASB operating at HRTs of 8-10 + 5-6 h achieved an average total and suspended COD removal efficiency of 55 and 62%, respectively. On the other hand, based on experimental results, it was calculated that a system AF + UASB working at HRTs of 4 + 8 h could provide a total COD removal efficiency higher than 70% under both summer and winter conditions. In the two-step UASB system most of the COD was removed in the first stage. It was calculated that a one-stage UASB reactor combined with a physical (-chemical) treatment unit (like an AF) for the removal of particulates in the effluent could reach total COD removal efficiencies as high as 93%. Mgana (2003) reported that results from a two-step UASB system treating sewage in a community on-site set up were encouraging under tropical conditions, especially because sludge washed out from the first reactor was entrapped in the second. However, more study was needed to reach steady state in the second step. A two-step anaerobic system followed by a settler and a posttreatment system consisting of a high-rate algal pond and a series of two maturation ponds was studied by El Hafiane (2003). An overall COD removal efficiency of 80% was obtained in the anaerobic steps at an HRT of 46 h.

Two-phase anaerobic digestion, meaning the separation of the nonmethanogenic and the methanogenic digestion phases in separate reactors, has been extensively studied in the past (Fan *et al.*, 1973; Ghosh and Pohland, 1974; Ghosh *et al.*, 1975; Ghosh and Klass, 1978; Ghosh, 1981; Girard *et al.*, 1986; Lin *et al.*, 1989; Ghosh, 1991; Shimizu *et al.*, 1993; Lin and Ouyang, 1993; Hernandez and Jenkins, 1994; Anderson *et al.*, 1994; Romli *et al.*, 1994; Fongsatitkul *et al.*, 1995; Beccari *et al.*, 1996). However, the application of two-stage systems to raw domestic sewage is a rather recent proposition (van Haandel and Lettinga, 1994; Wang, 1994; Sayed and Fergala, 1995; Fergala, 1995). It has to be remarked here that "two-phase" does not necessarily mean the same as "two-stage" or "two-step", as described above. There is some controversy as to whether a separate acidogenic reactor would be profitable or not in the overall efficiency of the process. Although a certain pre-acidification of the wastewater is certainly beneficial, there is clear evidence that a complete acidification is detrimental in several aspects (Lettinga and Hulshoff Pol, 1991).

Other processes

Other processes have also been used for the anaerobic treatment of sewage. Among these processes, it is worth mentioning the anaerobic filter (Young and McCarty, 1969), traditional anaerobic digesters currently in use in China (Yi-Zhang and Li-bin, 1988; Kuo-Cheng, 1988; Yao-Fu *et al.*, 1988), the anaerobic attached film expanded bed (AAFEB) system (Jewell *et al.*, 1981; Jewell, 1985), the polyurethane carrier reactor (Derycke and Verstraete, 1986), the anaerobic fluidized bed reactor (AFBR) (Yoda *et al.*, 1985; Sanz *et al.*, 1988; Sanz and Fernández-Polanco, 1989; 1990; Collivignarelli *et al.*, 1991), packed-bed reactors (Collivignarelli *et al.*, 1991), plug-flow reactors (Orozco, 1988; 1996), and modified anaerobic baffled reactors (ABR) (Yu and Anderson, 1996).

Posttreatment

Anaerobic sewage treatment systems generally fail to comply with COD discharge standards as that established by Council Directive 91/271/EEC on Urban Waste Water Treatment, dictated by the European Union Council of Ministers (1991) (125 mgCOD/L), or the guideline proposed by the World Health Organisation (WHO, 1989) for unrestricted irrigation (less than 1000 fecal coliform per 100 mL, and less than 1 helminth egg per L). Therefore, a posttreatment step is mandatory in most cases to remove remnant COD, fecal coliform (as an indicator of pathogenic microorganisms), helminth eggs, and even nitrogen and phosphorus when direct reuse is not feasible. Waste stabilization ponds (WSPs) are among the most efficient and cost-effective posttreatment methods available (van Haandel and Lettinga, 1994; Cavalcanti, 2003). Other posttreatment methods proposed in the literature are the biorotor system, or rotating biological contactor (Castillo *et al.*, 1997; Tawfik, 2002), integrated duckweed and stabilization pond system (van der Steen *et al.*, 1999), trickling filters (Chernicharo and Nascimento, 2001), the downflow hanging sponge reactor (Agrawal *et al.*, 1997; Machdar *et al.*, 2000; Uemura *et al.*, 2002), activated sludge (von Sperling *et al.*, 2001), a baffled pond system (von Sperling *et al.*, 2001), dissolved air flotation (Penetra *et al.*, 1999), sequential batch reactors (Torres and Foresti, 2001), submerged aerated biofilters (Gonçalves *et al.*, 1999), reed bed systems (Yu *et al.*, 1997), among others. Tawfik (2002) presented a comprehensive review of posttreatment systems for anaerobic effluents.

RESEARCH NEEDS

It was shown above that full-scale application of UASB systems for the anaerobic treatment of sewage is limited so far to regions with constant and relatively warm temperature conditions. The success of UASB reactors is highly dependent on the SRT (Chen and Hashimoto, 1980), which is a key factor determining the ultimate amount of hydrolysis, acidification, and methanogenesis in a UASB system at certain temperature conditions (Jewell, 1987; Miron *et al.*, 2000). The SRT should be long enough to provide sufficient methanogenic activity at the prevailing conditions. The SRT is determined by the loading rate, the fraction of SS in the influent, the removal of SS in the sludge bed, and the characteristics of the SS (biodegradability, composition, etc.). The effect of temperature on the different factors affecting the SRT is still not completely elucidated. Accumulation of SS may become significant at temperatures lower than 18°C due to very slow hydrolysis, forcing a reduction of the loading rate (de Man *et al.*, 1986; Elmitwalli, 2000). Accumulation of undegraded SS may induce a reduction in the methanogenic activity of the sludge, a deterioration of bacterial aggregates, and the formation of scum layers, leading to overloading of the reactor. When too long SRTs are necessary and, therefore, only low loading rates can be accommodated, a two-step system, and even additional sludge stabilization is to be considered, as discussed above (Mahmoud, 2002). Despite considerable work devoted to the elucidation of the mechanisms of SS removal and hydrolysis, both the physical and the biological processes in the first step of a two-stage anaerobic treatment need further research. The removal of SS will depend on factors like HRT, V_{up}, gas release, and sludge bed characteristics, and also on the characteristics of the SS themselves (mainly size and density, but also charge and formation of exopolymers). The particle size distribution of the incoming SS may be an important factor in the subsequent hydrolysis of the entrapped particles. Moreover, the hydrolysis rate will depend on the composition of the particles (fraction of lipids, proteins, and carbohydrates). A model describing a two-stage anaerobic sewage treatment system was proposed by Zakkour *et al.* (2001). Mathematical modeling of the system, including physical and biological processes can help to gain more insight into the process, and will certainly provide a rational basis for the adequate management of the SRT in UASB reactors. A model should also provide a basis for deciding for one- or two-stage anaerobic systems according to local sewage characteristics and environmental conditions. In two-stage systems, the optimal SRT may differ considerably from one stage to the other, mainly because of the large difference in incoming suspended solids. Mathematical modeling can be valuable in orienting future research on the subject, and can serve as a design tool for the development, transfer, and dissemination of anaerobic technology for direct sewage treatment in subtropical and temperate (developing) countries.

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Table 6. Application of upflow anaerobic reactors to sewage treatment. If not indicated otherwise, experiments were conducted in UASB reactors. Acronyms defined below.

Place	V (m^3)	T (°C)	Influent concentration (mg/L)			Inoculum	HRT (h)	Removal efficiency (%)			Start-up (months)	Period (months)	Reference (comments)
			COD	BOD (COD_{dis})	TSS			COD	BOD (COD_{dis})	TSS			
South Africa	0.008	20	500	(148)	NP	Active sludge	24	90	(49)	60-65	1	1	Pretorius, 1971
Netherlands	0.030	21	520-590	(73-75)	NP	DSS	9	57-79	(50-60)	30-70	NP	1	Lettinga *et al.*, 1983b
Netherlands	0.120	12-18	420-920	(55-95)	NP	DSS	32-40	48-70	(30-45)	90	NP	3	Lettinga *et al.*, 1983b
Netherlands	0.120	18-20	248-581	(163-376)	NP	GS	12	72	(62)	NP	NP	17	Lettinga *et al.*, 1983b
Netherlands	0.120	7-18	100-900	53-474	10-700*	GS	4-14	45-72	(38-59)	50-89	NP	12	de Man *et al.*, 1986
Netherlands	6	10-18	100-900	53-474	10-700*	GS	9-16	46-60	(42-48)	55-75	NP	12	de Man *et al.*, 1986
Netherlands	20	11-19	100-900 150-550	53-474 43-157	10-700* 50-400*	GS	6.2-18	31-49	(23-46)	NP	NP	12	de Man *et al.*, 1986
Colombia	64	25	267	95	NP	DCM	6-8	75-82	75-93	70-80	6	9	Louwe Kooijmans and van Velsen, 1986; Lettinga *et al.*, 1987
Netherlands	0.120	12-20	190-1180	(80-300)	NP	GS	7-8	30-75	(20-60)	NP	NP	NP	de Man *et al.*, 1988
Netherlands	0.116	12-20	150-600	(70-250)	NP	GS	2-3	NP	(20-60)	NP	NP	NP	de Man *et al*, 1988 (EGSB reactor)
Mexico	0.110	12-18	465	NP	154	AAS	12-18	65	NP	73	NP	>12	Monroy *et al.*, 1988
Brazil	0.120	19-28	627	357	376	None	4	74	78	72	4	9	Barbosa and Sant'Anna Jr., 1989
Italy	336	7-27	205-326	55-153	100-250	None	12-42	31-56	40-70[†]	55-80[†]	NP	12	Collivignarelli *et al.*, 1991; Maaskant *et al.*, 1991
India	1200	20-30	563	214	418	None	6	74	75	75	2.5	12	Draaijer *et al.*, 1992
Netherlands	120	>13	391	(291)	---	GS	2-7	16-34	(20-51)	None	NP	35	van der Last and Lettinga, 1992
Netherlands	205	16-19	391	(291)	---	Grown on sand	1.5-5.8	≅ 30	(≅ 40)	None	NP	33	van der Last and Lettinga, 1992 (EGSB reactor)
Colombia	35	NP	NP	NP	NP	NP	5-19	66-72	79-80	69-70	NP	48	Schellinkhout and Collazos, 1992
Netherlands	1.2	13.8	976	454	641 *	DSS	44.3	33	50	47.0*	NP	28	Bogte *et al.*, 1993 (UASB-septic-tank)
Netherlands	1.2	12.9	821	467	468 *	DSS	57.2	3.8	14.5	5.8*	NP	24	Bogte *et al.*, 1993 (UASB-septic-tank)
Netherlands	1.2	11.7	1716	640	1201 *	GS	102.5	60	50	77.1*	NP	13	Bogte *et al.*, 1993 (UASB-septic-tank)

Acronyms: V = Volume; T = Temperature; COD = chemical oxygen demand; BOD = biological oxygen demand; $_{dis}$ = dissolved; TSS = total suspended solids; HRT = hydraulic retention time; NP = not provided; DSS = digested sewage sludge; GS = granular sludge; DCM = digested cow manure; AAS = adapted aerobic sludge; DS = digested sludge; NAS = non-adapted sludge.

Footnotes: [a]: air temperature; *: expressed as COD; [†]: obtained at temperatures of 15-20°C, HRT of 12 h and upflow velocity (V_{up}) of 0.58 m/h.

Table 6 (continued).

Place	V (m^3)	T (°C)	Influent concentration (mg/L)			Inoculum	HRT (h)	Removal efficiency (%)			Start-up (months)	Period (months)	Reference (comments)
			COD	BOD (COD_{dis})	TSS			COD	BOD (COD_{dis})	TSS			
Indonesia	0.86	NP	NP	NP	NP	NP	360	90-93	92-95	93-97	NP	60	Lettinga *et al.*, 1993 (UASB-septic-tank, black water)
Indonesia	0.86	NP	NP	NP	NP	NP	34	67-77	up to 82	74-81	NP	60	Lettinga *et al.*, 1993 (UASB-septic-tank, grey + black water)
Thailand	0.030	30	450-750	NP	NP	Different types	3-12	90	NP	NP	>2	4	Gnanadipathy and Polprasert, 1993
Brazil	120	18-28	188-459	104-255	67-236	GS	5-15	60	70	70	>2	24	Vieira and Garcia Jr., 1994
Colombia	3360	24	380	160	240	None	5.0	45-60	64-78	≅ 60	>6	>36	Schellinkhout and Osorio, 1994
Brazil	67.5	16-23[a]	402	515	379	DS	7.0	74	80	87	NP	14	Vieira *et al.*, 1994
Netherlands	0.200	15.8	650	346	217	DS	3.0	37-38	26.6	83	None	5	Wang, 1994 (HUSB reactor)
Netherlands	0.120	15.8	397	254	33	GS	2.0	27-48	(32-58)	NP	None	3	Wang, 1994 (EGSB reactor)
Puerto Rico	0.059	≅20	782	352	393	DS	6-24	57.8	NP	76.9	≅4	16	Tang *et al.*, 1995
India	12000	18-32	1183	484	1000	NP	8	51-63	53-69	46-64	5	13	Haskoning, 1996; Tare *et al.*, 1997
India	6000	18-32	404	205	362	NP	8	62-72	65-71	70-78	5	11	Haskoning, 1996b; Tare *et al.*, 1997
Brazil	477	NP	600	NP	303	NAS	13	68	NP	76	2	>7	Chernicharo and Borges, 1997
Canada	0.008	20	350-500	(150-300)	100-270	DS	10-40	60-75	(70-85)	86	2	9	Singh and Viraraghavan, 1998
Brazil	9	NP (high)	712	312	386	UASB sludge	7.5	79	74	92	1	16	Chernicharo and Cardoso, 1999
Japan	0.021	13-25	312	(114)	NP	GS	4.7	69	(56)	NP	NP	6	Uemura and Harada (2000)
Netherlands	0.004	13	456	(112)	NP	GS	8	67	(30)	NP	1	2	Elmitwalli, 2000 (raw sewage)
Netherlands	0.004	13	339	(124)	NP	GS	8	60	(49)	NP	2	3	Elmitwalli, 2000 (settled sewage)
Brazil	810	30.8	549	(313)	196	Various	9.4	75	(73)	51	3	30	Florencio *et al.*, 2001
Netherlands	0.140	15	721	(172)	NP	UASB sludge	6	44	(5)	NP	1	3	Mahmoud , 2002
Jordan	60	18-25	1531	(277)	396	None	8-10	50	(-7)	41	NP	12	Halalsheh, 2002 (first stage of a two-stage UASB system)
Jordan	60	18-25	1531	(277)	396	None	23-27	51	(23)	55	NP	12	Halalsheh, 2002 (one-stage UASB reactor)
Brazil	1.5	NP	480	NP	NP	None	6	79	NP	NP	NP	NP	Cavalcanti, 2003
Tanzania	1.5	25-34	529	431	NP	Septic tank sludge	1.7-40	64	(64)	NP	6	42	Mgana, 2003

CHAPTER 2

Sewage characteristics in Salta, Argentina

ABSTRACT

The basic composition and characteristics of different types of sewage from the city of Salta (Argentina) were studied as the first step in the assessment of the feasibility of anaerobic treatment in the region. Raw sewage, with a total chemical oxygen demand (COD) of 214.8 ± 8.2 mg/L, can be classified as "very diluted". Total COD in settled sewage was 143.3 ± 6.8 mg/L. Mean sewage temperature in raw and settled sewage was 23.0 ± 0.3ºC. Temperature dropped below 20ºC for some months per year, with a minimum monthly average of 17.2 ± 0.7ºC, and a minimum daily average of 12.6 ± 2.4ºC. Total anaerobic biodegradability of raw sewage was approximately 70 and 65% at 30 and 20ºC, respectively. The anaerobic biodegradability of the suspended COD fraction of raw sewage was 86% at 30ºC and 76% at 20ºC. The anaerobic biodegradability of the colloidal and dissolved COD fractions grouped together was very low at 30ºC (approximately 25%) and no degradation of these fractions was observed at 20ºC. The first order hydrolysis rate constant (k_h) of organic solids present in raw sewage amounted to -0.21 ± 0.05 d^{-1} at 30ºC and -0.13 ± 0.04 d^{-1} at 20ºC (differences were significant at α = 0.05). Anaerobic biodegradability and k_h measured in concentrated sewage (raw sewage enriched with secondary sludge) were similar to the values obtained with raw sewage. Suspended COD in concentrated sewage was significantly higher than in raw sewage, but colloidal and dissolved COD were similar in both types of sewage. Judging by the sewage temperature regime over the year, its composition, and the anaerobic biodegradability of the organic pollutants, it can be concluded that raw, settled, and even concentrated sewage from the city of Salta are well-suited for anaerobic treatment in UASB reactors, at a design upflow velocity of 0.67 m/h (hydraulic retention time of 6 h in 4-m tall UASB reactors). With these hydraulic parameters, the solids retention time was estimated to be almost 300 d, more than enough to achieve sufficient hydrolysis and methanogenesis, a good treatement efficiency, and a satisfactory sludge stabilization.

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INTRODUCTION

The anaerobic degradation of complex organic waste can be subdivided in four main steps: hydrolysis, acidification, acidogenesis and methanogenesis (Sanders, 2001). Hydrolysis is the chemical degradation of the carbohydrates, proteins, and lipids present in suspended compounds and colloidal matter into their monomeric or dimeric components. Hydrolysis is generally considered the rate-limiting step during the anaerobic digestion of particulate organic matter (Pavlostathis and Giraldo-Gomez, 1991). First order kinetics are commonly applied to hydrolytic processes, although other models were proposed (Eastman and Ferguson, 1981; Sanders, 2001; Batstone *et al.*, 2002). Preliminary conversion mechanisms like cell lysis, non-enzymatic decay, phase separation, and physical breakdown, precede the more complex (chemical) hydrolytic steps when a composite organic material like primary sludge, anaerobic sludge, or raw sewage is degraded (Batstone *et al.*, 2002). As these processes are also assumed to be first order, lumped hydrolytic kinetic constants can be used (Batstone *et al.*, 2002). Hydrolysis rate constants have been determined in sewage sludge (Miron *et al.*, 2000; Mahmoud, 2002) and raw sewage (Halalsheh, 2002). A review of hydrolytic parameters was presented by Batstone *et al.* (2002).

The anaerobic biodegradability can be defined as the percentage of the chemical oxygen demand (COD) present in an organic sample that is transformed into methane under anaerobic conditions. The anaerobic biodegradability is also known as the percentage of methanogenesis. The anaerobic biodegradability is the anaerobic analogous of the biological oxygen demand (BOD) which in turn, represents the aerobic biodegradability of a sample. Within a certain range of temperature, the final anaerobic biodegradability is pretty constant, yet the degradation rate can vary considerably (Elmitwalli, 2000; Mahmoud, 2002). Results reported in literature should be compared with care because a standard biodegradability test is lacking.

Knowledge on the anaerobic biodegradability and the hydrolysis rate constant of a particular sewage can be a first indication of the potential applicability of anaerobic treatment. However, the performance of an anaerobic reactor under field conditions depends not only on biological processes, but also on the physical removal of suspended particles. Removal of suspended solids (and colloids) in sewage occurs by physical processes such as settling, adsorption, and entrapment. Subsequent hydrolysis and methanogenesis of the removed particulate fraction both depend mainly on temperature and the solids retention time (SRT). The lower the temperature the longer the SRT required in one-step UASB reactors to provide enough hydrolysis and methanogenesis to degrade the previously entrapped organic particulate fraction, assuming a certain removal efficiency of this fraction and a certain sludge concentration in the reactor (Zeeman *et al.*, 2001). A specific SRT is then required for each temperature and for each type of sewage. If this required SRT is known, based on literature or former experiences, the needed hydraulic retention time (HRT) can be calculated with the equation proposed by Zeeman and Lettinga (1999):

$$HRT = \left(\frac{C * SS}{X}\right) * R * (1 - H) * SRT \tag{1}$$

where HRT = hydraulic retention time (d); C = influent concentration (gCOD/L); SS = particulate fraction of COD influent (-); X = expected sludge concentration in the reactor (gCOD/$L_{reactor}$); R =

fraction of the particulate COD which is expected to be removed (-); H = fraction of removed particulates which will be hydrolyzed (-); and SRT = solids retention time (d).

Sewage temperature, composition, and anaerobic biodegradability are variable from place to place (Haskoning and WAU, 1994; Henze and Ledin, 2001; Mahmoud, 2002). Therefore, a complete characterization of sewage is advisable prior to the design of treatment facilities (Metcalf and Eddy, 1991).

The objective of this work was to characterize different types of sewage from the city of Salta (Argentina) as a first step in the assessment of the feasibility of anaerobic treatment in the region.

MATERIALS AND METHODS

Location

Argentina is a large country with an area of 3,761,274 km^2 (INDEC, 2001). It extends between 22° and 55°S and occupies the southern portion of South America, east of the crest-line of the Andes, which forms its border with Chile. On the north it is bordered by Bolivia and Paraguay and on the east by Brazil and Uruguay. From the estuary of the Río de la Plata to the southern tip of Tierra del Fuego its coastline is on the Atlantic Ocean. The country is mostly temperate but since the territory extends from further north of the Tropic of Capricorn to the South Pole there are many variations of weather and climate.

The population of the country was 36,260,130 inhabitants in the last census conducted in 2001, with 89.3% living in cities (INDEC, 2001). The annual rate of growth has been decreasing steadily from 3.6% in 1895 to 1.1% in 1991. Population density is 9.7 inh/km^2, but the capital city (Buenos Aires) and its surroundings hold more than 10 million inhabitants. The rest of the country has 14 cities with more than 100,000 inhabitants, some of them with over 500,000 inhabitants.

The city of Salta (population 462,051) is the capital of Salta province (population 1,079,051) (INDEC, 2001). The city is located in the northwestern part of Argentina, at 24º51'S 65º29'W and 1187 m above sea level. Mean ambient temperature in Salta, measured over a 22-y period (1971-1992) at a meteorological station close to the city, was 16.5 ± 0.2ºC (Arias and Bianchi, 1996). The climate in the surrounding region can be defined as subtropical with a dry season, although it can also be included within the zone of tropical climates, as an intermediate category between humid and dry climates (Martyn, 1992). In Salta, as in all northwest Argentina, cloudiness is maximum in summer and minimum in winter (Martyn, 1992). Below-zero minimum temperatures are possible some weeks per year during wintertime, especially at night. In Salta province, domestic sewage and rainwater are generally collected in separated sewer systems. Current sewage treatment plants are conventional systems like waste stabilization ponds (in several locations including the capital) and trickling filters (only in the capital).

Sewage

Sewage samples were obtained at the city's main sewage treatment plant, operated by a private company in charge of drinking water and sanitation in the whole province. The sewage treatment plant consists of preliminary treatment (screens and grit chamber), primary sedimentation, secondary treatment in trickling filters, secondary sedimentation, sludge digestion, and sludge drying. Hydraulic retention time in the settlers is about 2 h. Final disinfection of the effluent (chlorination) was suspended many years ago due to financial constraints. The sewage treatment plant is overloaded, as current sewage flow rate (more than 4500 m^3/h) exceeds the design value. During rainy periods, a varying amount of raw sewage diluted with rainwater is distracted from the plant, and discharged untreated into the receiving river (Río Arenales) through a by-pass pipe. With heavy rainfall up to 100% of the sewage is by-passed for several hours. Since November 2001, secondary sludge has been routinely recirculated into the incoming raw sewage. The objective of this procedure was to increase the organic load in the hydraulically overloaded trickling filters, especially during nighttime when sewage concentration was very low. This "concentrated" sewage was used in experiments described in chapter 5 of this thesis.

Sewage temperature

Temperature in settled sewage was measured twice a week during almost 8 years (August 1995 – March 2003) with a digital thermometer (Keithley). Since December 1999 to March 2003 settled sewage temperature was continuously monitored with a thermograph Novasen 3752-5-S-C (temperature range: 0-50ºC). The thermograph produced a continuous record in paper circles, which were replaced weekly. Hourly temperature readings were visually extracted from this continuous record and manually loaded on a computer worksheet. Daily means were calculated out of hourly readings. Raw sewage temperature was measured twice a week since November 2001 to March 2003 in grab samples. In this period, measurements in raw and settled sewage were performed with the same digital thermometer at the same times of the day.

Sample Analyses

Composite samples of raw, settled, and concentrated sewage were taken two to three times a week (0.5 L every 3 h for 24 h). Raw and concentrated sewage samples were taken after preliminary treatment units. Settled sewage was obtained after primary sedimentation. Samples were kept at 4ºC until they were analyzed. Total COD (COD_{tot}), paper-filtered COD (COD_{filt}) (Schleicher & Schuell 595½ 4.4-µm paper filters), and membrane-filtered (dissolved) COD (COD_{dis}) (Schleicher & Schuell ME 25 0.45-µm membrane) were determined in the samples. Suspended and colloidal COD (COD_{sus} and COD_{col}) were calculated as (COD_{tot} - COD_{filt}) and (COD_{filt} – COD_{dis}), respectively. BOD was determined in raw samples (before filtration). Total suspended solids (TSS) and volatile suspended solids (VSS) were determined in the residue retained by Schleicher & Schuell 589 4.4-µm ashless paper filters. Alkalinity was determined by titration with sulfuric acid to pH 4.5. VFA were determined with the distillation method. COD, BOD, and compounds of nitrogen, phosphorus, and sulfur were determined with HACH® micromethods. The rest of the analyses were performed according to APHA *et al.* (1995). Statistical comparisons and confidence intervals (CIs) were built at a level of significance (α) of 0.05 (5%).

Biodegradability tests

The total anaerobic biodegradability of raw and concentrated sewage was assessed in bench-scale batch experiments. Composite sewage samples (0.5 L every 3 h for 24 h) were anaerobically digested in duplicate 1-L sealed serum bottles. Anaerobic sludge was not added to the bottles as inoculum. Tests were performed at 20 and 30ºC using a refrigerator equipped with an Incutrol® incubator, and a temperature-controlled room, respectively. Methane production was monitored in time through the displacement of a 5% NaOH solution. The total anaerobic biodegradability was calculated as the amount of methane produced during the test (as COD) divided by the initial COD of the sample.

Combined hydrolysis and biodegradability tests

The hydrolysis rate constant was determined in batch experiments using sealed serum bottles where raw and concentrated sewage samples were anaerobically digested in the presence of viable and well-stabilized (granular) anaerobic sludge. The degree of hydrolysis was followed by measuring the production of methane. Methanogenic activity provided by the sludge added to the bottles (calculated with equation 9) was always in excess, and hydrolysis was therefore the rate-limiting step in the process of anaerobic digestion (Sanders, 2001). The effect of biomass growth on the value of the hydrolysis rate constant was neglected, according to Sanders (2001). The total anaerobic biodegradability of the pollutants was also calculated at the end of the tests.

Set up

Tests were performed at 20 and 30ºC. Methane was also measured by displacing a 5% NaOH solution from the gas collection vessel. The set up of the tests is described in **Table 1**. Macronutrients, trace elements, and phosphate buffer were added according to van Lier (1995) and Zehnder *et al.* (1980).

Raw or concentrated sewage from composite samples (2.5 L every 3 h for 24 h) was used in bottles 1 to 3 (blacks). Blanks (bottles 4 and 5) filled with distilled water were run in parallel to subtract the effect of methane production from the seed sludge. Bottles called "grays" (bottles 6 and 7) filled with filtered sewage (through Schleicher & Schuell 595½ 4.4-μm paper filters) were also incubated. In these bottles, methane is produced from the degradation of COD_{col}, COD_{dis}, and the sludge. Methane coming only from the degradation of COD_{sus} was calculated by subtracting the methane produced by the grays from that of the blacks.

Preliminary measurements indicated that methane produced only by the degradation of COD_{col} was too low to be distinguished in tests with sludge. Therefore, hydrolytic and biodegradability parameters were not measured for the colloidal COD fraction separately. Elmitwalli (2000) reported a high anaerobic biodegradability of COD_{col} at 30°C. However, the interest of this fraction during anaerobic treatment is limited, as its initial concentration in raw sewage is relatively low compared with the COD_{sus} fraction (particularly in the sewage from Salta) and its removal efficiency rarely exceeds 50% (Elmitwalli, 2000; Mahmoud, 2002).

Table 1. Hydrolysis and biodegradability test set up. V = bottle volume; MN = macronutrients; TE = trace elements; YE = yeast extract; PB = phosphate buffer; RS = raw sewage; DW = distilled water; FS = filtered sewage. Sludge was added according to equation 9.

Bottles	Name	V (L)	Substrate	Sludge (g)	MN (mL)	TE (mL)	YE (g)	PB (mL)	Measurement
1 to 3	Blacks	1	RS	Yes	2	2	0.2	10	Methane
4 – 5	Blanks	1	DW	Yes	2	2	0.2	10	Methane
6 – 7	Grays	1	FS	Yes	2	2	0.2	10	Methane

Calculations

Suspended organic matter in the bottles at any time during the anaerobic degradation was described by the following equation (Sanders, 2001):

$$COD_{sus(t)} = f_h COD_{sus(0)} \times e^{-k_h \cdot t} + (1 - f_h) COD_{sus(0)} \tag{2}$$

where $COD_{sus(t)}$ = biodegradable plus non biodegradable suspended organic matter at time t (gCOD/L); f_h = suspended biodegradable fraction (-) (varies between 0 and 1); $COD_{sus(0)}$ = biodegradable and non biodegradable suspended organic matter at the beginning of the test (time = 0) (gCOD/L); and k_h = hydrolysis rate constant for biodegradable suspended COD (d^{-1}). k_h was calculated as the slope of a straight line derived from equation 2:

$$\ln \frac{COD_{sus(t)} - (1 - f_h) COD_{sus(0)}}{f_h COD_{sus(0)}} = -k_h t \tag{3}$$

Term $COD_{sus(t)}$ in equation 3 was obtained as

$$COD_{\mathrm{sus}(t)} = COD_{sus(0)} - \text{Degraded}\, COD_{\mathrm{sus}(t)} \tag{4}$$

Degraded $COD_{sus(t)}$, f_h, the total percentage of hydrolysis, and the total percentage of methanogenesis (total anaerobic biodegradability) were calculated with equations 5, 6, 7, and 8, respectively (Veeken and Hamelers, 1999; Mahmoud, 2002). It's worth noticing that f_h refers solely to the fraction of suspended COD.

$$\text{Degraded}\, COD_{sus(t)} = T - G \tag{5}$$

$$f_h = \frac{T - G}{COD_{sus(0)}} \tag{6}$$

$$Hydrolysis\ (\%) = 100 \cdot \left(\frac{T - G}{COD_{tot(0)}} \right) \tag{7}$$

$$Methanogenesis\ (\%) = 100 \cdot \left(\frac{T - B}{COD_{tot(0)}} \right) \tag{8}$$

where T, G, and B = total amount of gaseous methane (gCOD/L) produced by blacks, grays, and blanks, respectively, at the end of the tests (t = ∞). Dissolved methane was not taken into account, because an identical amount was assumed to be present in all bottles (blacks, grays, and blanks).

Inoculum

Granular sludge from a UASB reactor treating settled sewage was used. The sludge was previously washed with distilled water in a kitchen filter to eliminate as much non-granular COD as possible.

Washed sludge was acclimatized (without substrate) at either 20 or 30ºC for 48 h before the tests started. The specific methanogenic activity (SMA) of intact and washed sludge was measured in parallel at both temperatures, according to the procedure used by van Lier (1995). As hydrolysis rate depends on the substrate concentration, the highest production of volatile fatty acids that have to be removed by methanogenic bacteria is expected in the first day of the test. Therefore, the minimum amount of inoculum that has to be added to treat the suspended fraction of the COD was estimated as follows (Sanders, 2001):

$$W = \frac{f_h COD_h (1 - e^{k_h \cdot t})}{SMA \cdot VSS \cdot t} \tag{9}$$

where W = sludge (g/L); COD_h = concentration of hydrolyzable substrate in sewage (gCOD/L); SMA = specific methanogenic activity of the sludge (gCOD-CH_4/gVSS.d); VSS = volatile suspended solids concentration in the sludge (gVSS/g of sludge); and t = 1 d. Values for the different variables initially used in this calculation (taken from previous measurements or from literature) were the following: f_h = 0.9, COD_h = 0.3, k_h = -0.15, SMA = 0.05, and VSS = 0.05. The calculated amount of sludge needed was 15 g/L. The sludge eventually added to the bottles was slightly higher than the calculated value to ensure an excess of methanogenic activity. At the end of each test, this calculation was done again with measured values for the different variables, to check if the added sludge was indeed enough to guarantee an excess of methanogenic activity.

RESULTS AND DISCUSSION

Sewage composition

Basic composition of raw, settled, and concentrated sewage is presented in **Table 2**. In terms of COD and BOD, raw sewage from the city of Salta can be classified as "very diluted", while concentrated sewage ranks between "moderate" and "diluted" (according to the categories in Henze and Ledin, 2001). Part of the original COD_{col} and COD_{dis} fractions may has been degraded in the sewerage system before reaching the treatment plant, located in the outskirts of the city (about 10 km from the center), explaining low concentrations observed for these fractions. Total and volatile suspended solids (TSS, VSS), and settleable solids in raw sewage are also close to values reported by Henze and Ledin (2001) for very diluted sewage. The low strength character of the sewage can be attributed to excessive water consumption, uncontrolled squandery, infiltration of rainwater through faulty sewer pipes, and illegal discharges of urban run-off directly into the sewerage. Estimations indicate that per capita domestic sewage production is higher than 350 L/inh.d, more than 200% higher than the value commonly observed in the Netherlands (Mels, 2001). Water use and, consequently, sewage production are expected to drop in the near future, as flow meters are currently being installed in the city to monitor and charge household consumption. Sewage concentration is due to rise accordingly. A more concentrated raw sewage can be expected in the rest of the province, where water consumption is lower than in the capital.

The ratio COD_{sus}/COD_{tot} was 52% in raw sewage, 44% in settled sewage, and 72% in concentrated sewage. COD_{tot} in settled sewage did not change significantly after recirculation of secondary sludge started, going from 151.9 ± 5.3 g/L before recirculation (data from more than two years) to 143.3 ± 6.8 g/L after recirculation (see **Table 2**). COD_{tot} removal efficiency increased in primary settlers from 33% before recirculation to 67% after recirculation. These figures make clear that recirculated sludge did not eventually reach the trickling filters as planned, making the usefulness of the recirculation practices questionable. COD_{col} represented 12% of the COD_{tot} in raw sewage, lower than the 20-30% proportion cited by Elmitwalli (2000) for the sewage from Bennekom (the Netherlands). The ratio BOD/COD indicates that about 60% of the COD in raw sewage is potentially biodegradable under aerobic conditions. Solids in concentrated sewage were significantly higher than in raw sewage. Alkalinity was relatively high and constant in all types of sewage. Volatile fatty acids (VFA) accounted for 50.2, 36.1, and 40.0% of COD_{dis} in raw, settled, and concentrated sewage, respectively. In general, nutrients were close to the values reported by Henze and Ledin (2001) for domestic sewage.

Table 2. Basic composition of different types of sewage from Salta[a]. ND = not determined.

Parameter	Raw sewage[b]	Settled sewage	Concentrated sewage
Year	2001	2002	2002
pH	7.57 ± 0.07 (12)	7.71 ± 0.08 (31)	7.62 ± 0,08 (32)
COD_{tot} (mg/L)	214.8 ± 8.2 (183)	143.3 ± 6.8 (75)	432.7 ± 36,9 (77)
COD_{sus} (mg/L)	111.9 ± 10.6 (83)	63.7 ± 6.1 (38)	311.9 ± 39,4 (40)
COD_{col} (mg/L)	25.7 ± 3.9 (84)	22.9 ± 4.9 (38)	33.4 ± 8,9 (39)
COD_{dis} (mg/L)	60.0 ± 4.1 (86)	51.5 ± 6.8 (38)	49.8 ± 6,4 (39)
BOD (mg/L)	127.7 ± 5.1 (129)	ND	ND
TSS (mg/L)	140.0 ± 4.6 (12)	37.4 ± 6.5 (36)	419.2 ± 76,8 (33)
VSS (mg/L)	98.5 ± 3.8 (12)	11.1 ± 2.2 (35)	207.7 ± 42,6 (32)
Settleable Solids in 2 h (mL/L)	2.1 ± 0.1 (12)	0.2 ± 0.1 (25)	6.7 ± 1,2 (26)
Alkalinity (mg $CaCO_3$/L to pH 4.5)	165.0 ± 16.1 (89)	175.7 ± 4.9 (31)	181.3 ± 6,4 (32)
VFA (mg of acetic acid/L)	30.1 ± 4.0 (86)	18.6 ± 1.9 (31)	19.9 ± 2,1 (31)
NO^{3-} (mg N/L)	ND	0.94 ± 0.19 (4)	1.56 ± 1.51 (3)
NO^{2-} (mg N/L)	ND	0.05 ± 0.04 (3)	0.01 ± 0.02 (3)
NH_4^+ (mg N/L)	ND	25.8 ± 4.5 (3)	19.9 ± 3.6 (3)
PO_4^{3-} (mg P/L)	ND	5.4 ± 2.5 (4)	5.5 ± 4.3 (3)
SO_4^{2-} (mg/L)	ND	41.0 ± 6.0 (3)	38.0 ± 4.0 (4)
S^{2-} (mg/L)	ND	0.09 ± 0.03 (2)	0.12 ± 0.14 (3)

[a] Mean values ± 95% CIs are provided (sample size between brackets).
[b] Some of the data were provided by Aguas de Salta S.A.

During the first 3 months of secondary sludge recirculation, samples of raw and concentrated sewage were taken and analyzed in parallel to find out the proportion of raw sewage and secondary sludge present in concentrated sewage (**Table 3**). COD_{col} and COD_{dis} were very similar in both types of sewage. Assuming that secondary sludge contributed only with COD_{sus}, it can be calculated that, in terms of COD, concentrated sewage was roughly composed of 40% secondary sludge and 60% raw sewage.

Table 3. Raw and concentrated sewage during the first three months of secondary sludge recirculation[a]. Sludge COD_{sus} was calculated as the difference between COD_{sus} in concentrated sewage and raw sewage.

COD fraction (mg/L)	Raw sewage	Concentrated sewage	Sludge (calculated)
COD_{tot}	212.7 ± 24.1	354.8 ± 38.8	
COD_{sus}	130.1 ± 31.4	261.6 ± 50.0	131.5
COD_{col}	25.8 ± 9.4	21.6 ± 6.2	
COD_{dis}	47.7 ± 7.4	52.1 ± 9.4	

[a] Mean values ± 95% CIs are provided.

Sewage temperature

More than two years of continuous temperature measurements in settled sewage are shown in **Figure 1**. Mean temperature during this period was 23.0 ± 0.3ºC. Extreme values observed were 12.6 ± 2.4ºC (daily minimum), 17.2 ± 0.7ºC (monthly minimum), 30.0 ± 2.5 (daily maximum), and 26.6 ± 0.7ºC (monthly maximum). Hourly readings below and above these values were recorded. A difference of about 2ºC was observed in mean temperature between the two years. After more than one year of parallel grab measurements, it was observed that raw and settled sewage have almost the same average temperature (the difference was 0.05 ± 0.09ºC, n = 98). Sewage from the city of Salta seems to be, on average, warm enough to be treated anaerobically. However, the situation in winter has to be taken into account because sewage temperature can drop below 20ºC for some months. Sewage temperature was on average 6.5ºC higher than ambient temperature. When it comes to sewage treatment, terms like "subtropical" or "moderate temperatures" should always be related to long-term sewage temperature measurements (at least one full year).

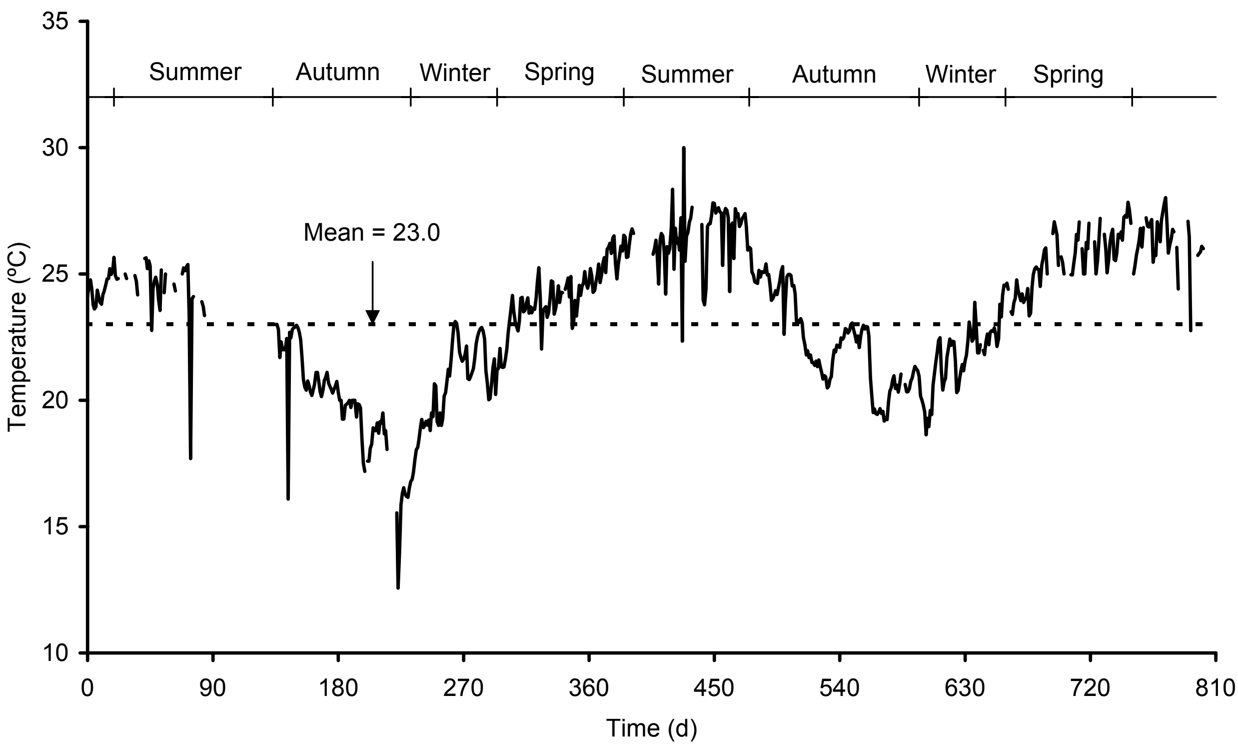

Figure 1. Settled sewage temperature in Salta. Daily means are shown, calculated out of hourly readings from a continuous record. Seasons indicated on top correspond to years 2000 and 2001.

Biodegradability

In raw sewage, total anaerobic biodegradability (measured through the percentage of methanogenesis) was similar in tests performed with and without sludge (**Table 4**). On the other hand, differences observed between both temperatures were not significant (**Table 4**). Results were quite close to values reported in literature. According to Henze and Ledin (2001) around 76% of

COD_{tot} is degradable for all types of sewage (from concentrated to very diluted). Elmitwalli (2000) reported that the total anaerobic biodegradability of raw sewage from Bennekom, the Netherlands, was 74% either at 20 and 30ºC, after 135 and 80 d of digestion, respectively. Halalsheh (2002) reported biodegradability ranging from 76 to 79% at 30ºC for different sewage sources in Jordan, in tests lasting from 130 to 224 d. Sewage from university facilities in Dar es salaam, Tanzania, showed anaerobic biodegradability of 64% (Mgana, 2003).

In our study, total anaerobic biodegradability in concentrated sewage was close to values observed in raw sewage, except for the test without sludge at 20ºC in which a low percentage of methanogenesis was observed for concentrated sewage. The duration of some of the tests at 20°C may have been too short to reach the ultimate biodegradability values. More tests are needed to confirm these results. Differences observed among tests with and without sludge were higher for concentrated sewage than for raw sewage.

Both types of sewage showed similar f_h values at 30ºC (**Table 4**). A higher f_h was observed in concentrated sewage than in raw sewage at 20ºC. Concentrated sewage was expected to have lower f_h values than raw sewage because secondary sludge, as the one present in concentrated sewage, is generally less biodegradable than suspended solids from raw sewage or primary sludge (Metcalf and Eddy, 1991). However, as the sewage treatment plant was overloaded, it is possible that remaining suspended solids from raw sewage constituted an important fraction of secondary sludge. In fact, settled sewage still contained almost 60% of the COD_{sus} from raw sewage (see **Table 2**).

The anaerobic biodegradability of COD_{col} and COD_{dis} could not be measured separately in these tests. However, knowing the percentage of methanogenesis and the f_h, undegraded COD_{tot} and undegraded COD_{sus} can be calculated. The difference between these two values is the amount of undegraded COD_{col} and COD_{dis} grouped together ($COD_{col+dis}$). Degraded $COD_{col+dis}$ would then be the initial $COD_{col+dis}$ added to the bottles minus the undegraded $COD_{col+dis}$. Therefore, the anaerobic biodegradability of $COD_{col+dis}$ can be calculated as degraded $COD_{col+dis}$ divided by initial $COD_{col+dis}$. Anaerobic biodegradability of $COD_{col+dis}$ was 26.1% at 30ºC in raw and concentrated sewage. No degradation at all of these fractions was observed at 20ºC. Degraded $COD_{col+dis}$ was even lower than the initial VFA concentration, assumed to be about 50% of COD_{dis} (from data in **Table 2**). This means that VFA were not completely degraded during the tests. However, it has to be noticed that the distillation method used to determine VFA is empirical and can give incomplete and somewhat variable results, although it's a method suitable for routine control purposes (APHA *et al.*, 1995). It is probable that both COD_{col} and COD_{dis} were partially degraded, yet the exact proportions could not be determined with the experimental design used in these tests.

Hydrolysis

The assessed values for the mean hydrolytic parameters were similar for raw and concentrated sewage (**Table 4**). Paired comparisons of means showed that k_h was significantly higher at 30 than at 20ºC. The k_h observed at 30ºC was similar to that reported at 35ºC by Mahmoud (2002) for primary sludge digested for 10 to 30 d in a continuously stirred tank reactor. The k_h for primary sewage sludge could be regarded as an indicator of k_h for sewage, as sewage sludge is formed by settling of suspended solids from sewage. The assessed k_h values were higher than those reported by Halalsheh (2002) for the carbohydrate, protein, and lipid fractions of raw sewage at 15 and 25ºC.

Table 4. Hydrolysis and biodegradability parameters in raw and concentrated sewage[a]. A = without seed sludge; B = with seed sludge.

Parameter	Raw sewage				Concentrated sewage			
	30ºC		20ºC		30ºC		20ºC	
	A	B	A	B	A	B	A	B
COD_{tot} (mg/L)	299.0	259.8 ± 34.8	299.0	259.8 ± 34.8	383.5	375.8 ± 15.2	383.5	375.8 ± 15.2
COD_{sus} (mg/L)		222.0 ± 8.8		222.0 ± 8.8		282.5 ± 7.8		282.5 ± 7.8
COD_{col} (mg/L)		18.8 ± 4.4		18.8 ± 4.4		19.5 ± 7.8		19.5 ± 7.8
COD_{dis} (mg/L)		19.0 ± 30.4		19.0 ± 30.4		73.8 ± 15.2		73.8 ± 15.2
k_h (d^{-1})		-0.21 ± 0.05		-0.13 ± 0.04		-0.18 ± 0.08		-0.13 ± 0.03
f_h (-)		0.86		0.76		0.85		0.87
Hydrolysis (%)		73.8		65.5		72.3		68.0
Methanogenesis (%)	71.2	69.9	64.0	65.9	66.7	76.7	50.3	65.4
Test length (d)	93.9	46.4	93.9	51.6	81.7	15.0	81.7	33.5

[a]Mean values ± 95% CIs are presented except when only one value was available.

Cumulative methane production in blacks was always significantly higher than in grays, allowing an accurate determination of hydrolysis and biodegradability of COD_{sus} (see example from one test with raw sewage in **Figure 2**). Small differences were observed in methane production between grays and blanks. For that reason, a separate determination of hydrolysis and biodegradability of COD_{col} can be subject to a high degree of experimental error. Morever, COD_{col} represented only a fraction of the COD present in the grays, the rest being COD_{dis} (see values in **Table 2**). No significant lag phase was observed in the tests at both temperatures (**Figure 2**), showing that methanogenic activity in the bottles was indeed enough to remove soluble biodegradable compounds.

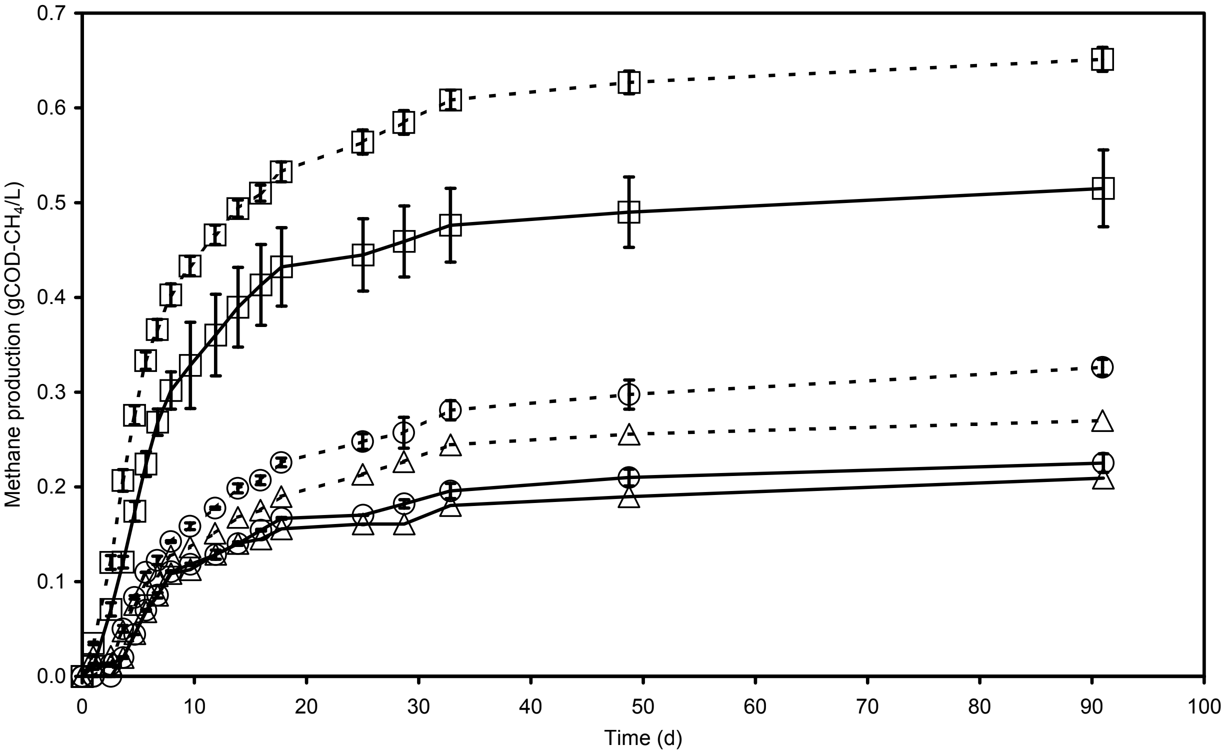

Figure 2. Methane production during a hydrolysis and biodegradability batch test conducted with raw sewage. Dotted line = test at 30ºC; full line = test at 20ºC; □ = raw sewage (blacks); O = filtered sewage (grays); Δ = water (blank). CIs (α = 0.05) are shown as Y error bars for blacks and grays.

However, as the first methane measurements were taken approximately 24 h after the incubation started, short lag phases could have been overlooked. Regular measurements of the concentration of dissolved COD and VFA were initially scheduled to check the assumption that hydrolysis was rate limiting. However, representative sampling from the bottles was very difficult and results from the analyses were inconsistent, probably because of the presence of sludge. SMA (at 30°C) of the washed anaerobic sludge used as inoculum was 0.083 gCOD-CH_4/gVSS.d, similar to that of intact sludge measured simultaneously (0.098 gCOD-CH_4/gVSS.d).

FINAL DISCUSSION

Unseeded biodegradability tests are the simplest way of measuring the total anaerobic biodegradability of a sewage sample. These tests are reliable and show low variability between duplicates. However, the time needed to complete a test is very long (3 months or more). Moreover, hydrolytic parameters can not be obtained in unseeded tests. The hydrolysis rate constant can only be measured when there is enough methanogenic activity in the bottles to guarantee that all by-products from hydrolysis are effectively removed, and hydrolysis can be assumed to be the rate-limiting step in the process of anaerobic digestion. Measurements of COD_{dis} and VFA during the digestion, as recommended by Sanders (2001) to check this assumption, are difficult to accomplish in tests with sludge, and hydrolysis results have to be based solely on methane production. The time needed to measure the total anaerobic biodegradability of sewage is shorter in tests with sludge than in unseeded tests. However, tests with sludge are slightly more complex and the variability between duplicates is higher due to the presence of anaerobic sludge. Tests with sludge may be useful to get a (relatively) fast first idea of the biodegradability of a sewage sample, while tests without sludge can be performed when a more accurate value is needed. If the hydrolysis rate constant needs to be known, then tests with sludge must be conducted.

In this study, equation 1 was used to calculate the HRT and the upflow velocity (V_{up}) needed for the application of a one-stage UASB reactor for the treatment of different types of sewage under the conditions prevailing in the city of Salta. **Figure 3** shows V_{up} (calculated for 4-m tall reactors) as a function of SRT for raw, settled, and concentrated sewage, as calculated on the basis of equation 1. Input values used to build this figure were the following: C = 214.8, 143.3, and 432.7 mgCOD/L for raw, settled and concentrated sewage, respectively (from **Table 2**); SS = 0.52, 0.44, 0.72 for raw, settled and concentrated sewage, respectively (calculated as COD_{sus}/COD_{tot} from data in **Table 2**; COD_{col} was neglected); X = 21 gCOD/$L_{reactor}$ (after Mahmoud, 2002); R = 0.8 (after Mahmoud, 2002); H = 0.8 (rounded value based on f_h reported in **Table 4**).

For a safe reactor operation, it can be assumed that the minimum acceptable SRT is 50 d (Haskoning and WAU, 1994; Cavalcanti, 2003), and the maximum acceptable V_{up} is 1 m/h (HRT = 4) although higher values may be acceptable for short periods of time (van Haandel and Lettinga, 1994). From **Figure 3** it is clear that the SRT will be adequate for all types of sewage, as long as the V_{up} is lower than 1 m/h (upper horizontal dotted line). On the other hand, an SRT of 50 d (vertical dotted line) would render unacceptably high values for V_{up}. An HRT of 6 h is generally recommended for the design of UASB reactors under tropical conditions, aiming at an V_{up} of about 0.67 m/h in 4-m tall reactors (van Haandel and Lettinga, 1994). At this V_{up} (indicated by the lower horizontal dotted line in **Figure 3**), calculated SRTs would be 105, 293, and 515 d for concentrated, raw, and settled sewage, respectively. These values are more than enough to achieve sufficient hydrolysis and methanogenesis, and a satisfactory sludge stabilization under local conditions. Based on these calculations, it can be concluded that the V_{up} is the limiting operational variable to be used as the main criteria for the design of UASB reactors for sewage treatment, in agreement with Haskoning and WAU (1994) and Halalsheh (2002). On the contrary, Cavalcanti (2003) concluded that the SRT, and not the V_{up}, was the limiting variable in the operation of UASB reactors for sewage treatment under her specific (i.e. tropical) conditions. As a rule, design has to be based on the SRT if hydrolysis is not sufficient to degrade the particles retained in the reactor. The rate of hydrolysis and the efficiency of the anaerobic process are highly dependent on the reactor temperature (van Haandel and Lettinga, 1994; Bogte et al., 1993; Veeken and Hamelers, 1999). In the case of sewage, term H is probably the most important parameter in equation 1. A small change in the hydrolysis of particulates leads to a large variation in the calculated SRT.

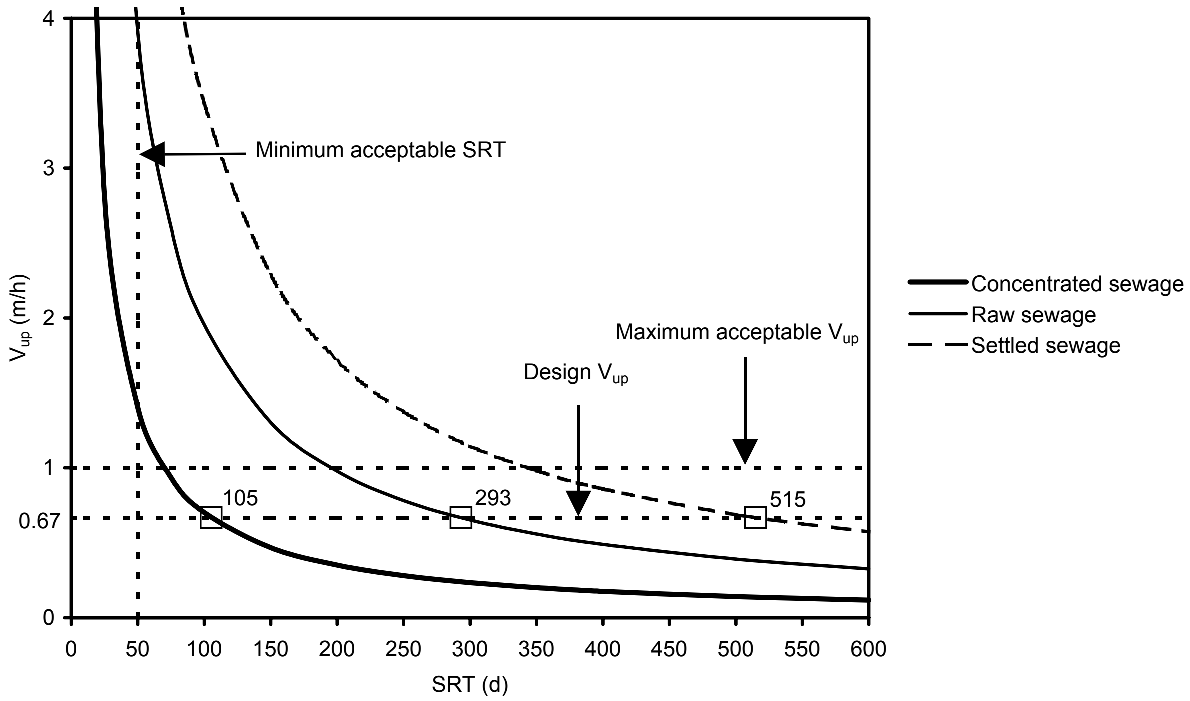

Figure 3. V_{up} required in 4-m tall UASB reactors for the treatment of different types of sewage as a function of SRT calculated with the equation proposed by Zeeman and Lettinga (1999).

Concentrated sewage represents the worst possible scenario for anaerobic treatment due to the high proportion of suspended solids. However, from **Figure 3** it can be deduced that even concentrated sewage could be treated in a single UASB reactor under local conditions. Experiments with laboratory or pilot-scale UASB reactors fed with specific types of sewage under well-defined environmental and hydraulic conditions are necessary to determine: (a) the physical behavior of the particulate fraction; (b) the amount of sludge that can be retained in the reactor; and (c) the characteristics of the sludge (COD and SS concentration, SMA, stability). In those experiments, not only terms R and X in equation 1 can be adjusted, but also a real SRT can be measured for the specific conditions applied.

CONCLUSIONS

- Raw sewage from the city of Salta, with a COD_{tot} concentration of 214.8 ± 8.2 mg/L, can be classified as “very diluted”. COD_{tot} concentration in settled sewage was 143.3 ± 6.8 mg/L.
- Mean sewage temperature in raw and settled sewage was 23.0 ± 0.3ºC. Temperature dropped below 20ºC for some months per year, with a minimum monthly average of 17.2 ± 0.7ºC, and a minimum daily average of 12.6 ± 2.4ºC.

- Total anaerobic biodegradability of raw sewage was approximately 70 and 65% at 30 and 20ºC, respectively. Anaerobic biodegradability of COD_{sus} in raw sewage was 86% at 30ºC and 76% at 20ºC. Anaerobic biodegradability of COD_{col} and COD_{dis} grouped together was very low at 30ºC (approximately 25%), and no degradation of these fractions was observed at 20ºC.
- The k_h in raw sewage was –0.21 ± 0.05 d^{-1} at 30ºC and –0.13 ± 0.04 d^{-1} at 20ºC (differences were significant at $\alpha = 0.05$).
- Anaerobic biodegradability and k_h measured in concentrated sewage (raw sewage enriched with secondary sludge) were similar to those obtained with raw sewage. COD_{sus} in concentrated sewage was significantly higher than in raw sewage, but COD_{col} and COD_{dis} were similar in both types of sewage.
- Judging by temperature, composition, and anaerobic biodegradability, it can be concluded that raw, settled, and even concentrated sewage from the city of Salta are well-suited for anaerobic treatment in UASB reactors.
- A design V_{up} of about 0.67 m/h (HRT = 6 h in 4-m tall UASB reactors) seems adequate for the anaerobic treatment of raw sewage in the region. The SRT calculated for these hydraulic parameters is expected to be almost 300 d, more than enough to achieve the required hydrolysis and methanogenesis under local conditions.

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CHAPTER 3

Anaerobic treatment of settled sewage under subtropical conditions in an upflow anaerobic sludge bed (UASB) reactor

ABSTRACT

The performance of a sewage treatment system consisting of a settler followed by a pilot-scale (0.501 m^3) upflow anaerobic sludge bed (UASB) reactor is presented. The system was in operation for more than seven years in the city of Salta (Argentina). It was the first experiment ever conducted in the country on anaerobic sewage treatment. The climate in the region is defined as subtropical with a dry season, with a mean ambient temperature of 16.5ºC. Mean annual sewage temperature was 23.0ºC, with up to four months in a row below 20ºC. A maximum total chemical oxygen demand (COD_{tot}) removal efficiency of 63.2% was observed in the reactor at a mean upflow velocity (V_{up}) of 0.42 m/h and a mean hydraulic retention time (HRT) of 6.1 h. Removal efficiencies up to 84.0% in total COD and 92.0% in suspended COD have been observed in the entire system at a mean HRT of 2 h in the settler and 5.6 h in the reactor (V_{up} = 0.71 m/h). The final effluent was in compliance with discharge standards of 125 $mgCOD_{tot}$/L. Semi-digested sewage sludge was used as inoculum, and a granular sludge bed developed in the reactor. Under steady state conditions, sludge growth rate represented 2.7% of the reactor volume per month, and the solids retention time (SRT) was 498 d. The mean volatile suspended solids (VSS) concentration in the sludge was 29 g/L, with a specific methanogenic activity (SMA) of 0.10 gCOD-CH_4/gVSS.d. Sludge characteristics (VSS concentration and SMA) were relatively constant in time despite variations applied in hydraulic conditions.

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INTRODUCTION

The upflow anaerobic sludge blanket reactor (UASB) is widely applied for sewage treatment in tropical countries. Its application in subtropical and temperate regions has been very limited so far. As discussed in chapters 1 and 2, hydrolysis, the rate-limiting step in the process of anaerobic digestion, may become too slow at low temperatures, inducing accumulation of total and volatile suspended solids (TSS and VSS), which may lead to deterioration of sludge quality, sludge flotation, and reductions in treatment efficiency.

Pre-removal of suspended solids may be needed when sewage is treated in UASB reactors at short hydraulic retention time (HRT) and low temperature (Elmitwalli, 2000). The application of two-stage anaerobic systems also seems to be promising to overcome the problem of solids accumulation under these conditions (van Haandel and Lettinga, 1994; Wang, 1994; Sayed and Fergala, 1995). In two-stage systems, the first stage retains most of the particulate fraction, which can then undergo a partial hydrolysis. The second stage further improves the removal of suspended solids and degrades newly formed soluble compounds, eventually leading to the formation of a granular sludge bed (van Haandel and Lettinga, 1994; Wang, 1994). In fact, granulation was observed in reactors treating settled sewage (van der Last and Lettinga, 1992) and pre-hydrolyzed sewage supplemented with sucrose (Ligero and Soto, 2002). Different first-stage reactors have been designed to retain and hydrolyze the particulate fraction (Wang, 1994; Sayed and Fergala, 1995; Zeeman *et al.*, 1997; Elmitwalli *et al.*, 1999).

Granulation of anaerobic sludge in UASB reactors is a widely known phenomenon observed mainly on soluble types of wastewater (Hulshoff Pol, 1989; Alphenaar, 1994). According to van Haandel and Lettinga (1994), it had not been observed in any of the existing full-scale UASB reactors treating sewage. However, Barbosa and Sant´Anna (1989) reported granulation in a 120-L UASB reactor treating raw sewage at temperatures ranging from 18 to 28ºC, at an HRT of only 4 h and an upflow velocity (V_{up}) of 0.48 m/h. Development of highly active granular sludge on settled sewage was observed at temperatures above 13ºC in a fluidized bed reactor seeded with silversand (Van der Last and Lettinga, 1992). Singh and Viraraghavan (1998) also reported biomass aggregation in the form of flocs and small granules when treating raw sewage at 20ºC. In their reactor, about 50% of the sludge were particles with a mean diameter of about 1.5 mm although the shape of the particles was not very regular.

The availability of primary sedimentation tanks (settlers) from conventional wastewater treatment plants provides settled sewage that can, in principle, be used as influent for experimental "second-stage" UASB reactors. Wang (1994) showed that up-flow anaerobic reactors are more efficient than settlers in removing organic matter, expressed as chemical oxygen demand (COD) and suspended solids. His two-step anaerobic system provided an average COD removal of 71% at temperatures over 15ºC. Based on this, it's reasonable to expect that two-step UASB systems would remove more COD than a system Settler + UASB.

Anaerobic sludge is often characterized by its specific methanogenic activity (SMA). SMA determination is useful to select seed sludges, to determine potential organic loading rates applicable (when suspended solids concentration in the influent is low), to follow the development of the sludge, to detect inhibitions and toxic effects, to prevent accumulation of inert material in the sludge bed, and to determine the sludge activity profile within anaerobic reactors (Soto *et al.*, 1993). SMA is measured in batch tests where a fixed amount of sludge is fed with an excess of biodegradable substrate (e.g. acetic acid) under optimum environmental conditions. SMA is then calculated from the maximum methane production rate. SMA in granular sludge was found to be inversely related to the size of the granules because internal mass transfer limitations are more important than external mass transfer limitations in anaerobic granular sludge (Alphenaar, 1994). The larger the granules, the higher the Monod´s apparent half-saturation constant (K_s), a parameter linked to the sludge affinity for the substrate (González-Gil *et al.*, 2001). The SMA indicates the maximum potential of a reactor to produce methane and can give an indication of the sludge variability along the reactor height. As sludge withdrawal has to be performed periodically in anaerobic reactors (Cavalcanti *et al.*, 1999), the SMA profile could be a useful tool for sludge management. The least active sludge should be disposed of (provided that it is well stabilized) while the most active layers could be used as inoculum for new reactors. For management purposes, the density of the sludge has to be taken into account as well, especially when the sludge can not be dried on the spot or has to be transported over long distances before further reuse or final disposal.

Hydraulic retention time (HRT) and V_{up} may influence the distribution of the sludge, eventually segregating layers based on density differences or concentration gradients (Alphenaar *et al.*, 1993). HRT also determines the solids retention time (SRT) that can be reached in the reactor (Zeeman and Lettinga, 1999). V_{up} may influence granulation as a result of washout of influent suspended solids. The relationship between V_{up} and COD removal efficiency is not completely clear in full-scale UASB reactors (Wiegant, 2001).

The objectives of this work were: (a) to study the long-term performance of a sewage treatment system consisting of a settler followed by a UASB reactor; (b) to determine SMA and VSS profiles over the sludge bed of the UASB reactor; and (c) to study the effect of V_{up} on COD removal efficiency and sludge characteristics (SMA and VSS concentration) in the UASB reactor.

MATERIALS AND METHODS

Pilot Plant

Experiments were performed in Salta, Argentina. The climate in the region is defined as subtropical with a dry season. Ambient temperature in the city and surroundings is 16.5 ± 0.2ºC (Arias and Bianchi, 1996). See chapter 2 for a more detailed description of the location.

The pilot plant (**Figure 1**) was installed at the city's main sewage treatment plant described in chapter 2. Preliminary treatment was provided by screens (retention of coarse materials) and grit chamber (sand trap). The settler had a volume of 4400 m^3 (diameter = 42 m; height at the edge = 2.90 m; total height in the center = 4.40 m; sludge channel height = 2.60; sludge channel width = 4.50 m). The UASB reactor had a working volume of 0.501 m^3 (height = 2.55 m; diameter = 0.50 m). In the last stage of the research, reactor height was increased to 3.95 m as recommended for full-scale UASB reactors (van Haandel and Lettinga, 1994; Haskoning and WAU, 1994). Total volume after this change was 0.776 m^3. Fifteen sampling ports (diameter = 3/4 in.) separated 0.15 m from each other were installed along the reactor for liquid and sludge sampling. Polyvinyl chloride (PVC) tubes and hoses (internal diameter = 1 in.) were used for influent and effluent distribution. A retention valve was installed at the influent entrance to prevent sludge from flowing out during maintenance operations and power cuts. The influent was distributed in the reactor through one inverted inlet pipe located 5 cm from the bottom. The reactor and the biogas accumulator were constructed of polyester reinforced with glass fiber by a local company (JJS Industrias Plásticas y Mecánicas).

Sewage was pumped with a peristaltic pump Watson Marlow 601 F/R Close Couple (flow rate range: 6–960 L/h) equipped with Marprene® tubing. Pump speed could be electronically changed with an Adjustable Frequency Drive VLT®MICRO Danfoss (1/2 HP; 0.4 kW). Flow rate was measured with a Kobold KSK 3500 flow meter (measuring range: 0.83–8.30 L/min).

The biogas was led to an accumulation system and automatically measured in a domestic gas meter (Galileo MGD G2D1) operated by electric valves (Jefferson) and switches (Neumann CB 130). Methane content in the biogas was determined by displacing a 5% NaOH solution from a tightly closed, up side down serum bottle into a graduated cylinder. CO_2 was retained in the solution. The content of other gases in the biogas, like hydrogen sulfide, was neglected.

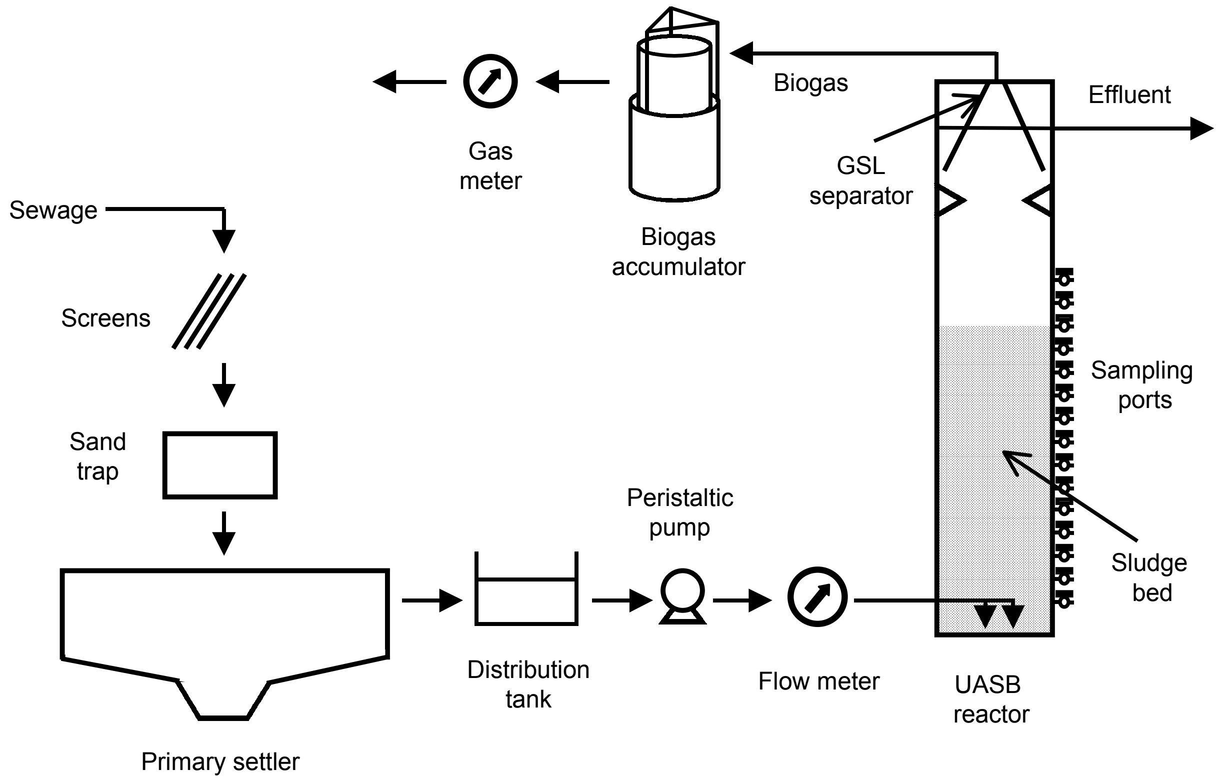

Figure 1. Schematic diagram of the system studied. Description of the equipment provided in the text. GSL = gas-solid-liquid. Not to scale.

Experimental set up

The system was designed to operate at HRTs of 2 h in the settler, and 3 to 24 h in the UASB reactor. The treatment plant's employees operated the settler and we had no control over its operation. About 100 L of partially-digested primary sewage sludge (20% [v/v]) were used to inoculate the UASB reactor. Feeding with settled sewage started 24 h after inoculation at an HRT of 14 h (V_{up} = 0.18 m/h). During start up, the HRT was rapidly decreased to about 9 h. Operation was divided in periods. A detailed description of the operation during the whole experiment is presented in **Table 2** (left half). Period 0 included the startup and the first 1481 d (4.1 y) of operation (HRT = 8.2 ± 0.5 h; V_{up} = 0.31 ± 0.02 m/h; n = 277). Details about period 0 have not been included mainly because influent flow rate control in the UASB reactor was not completely reliable (the peristaltic pump had not been installed yet) and the collection of data was less frequent than in subsequent periods. Period 1 includes more than 2 y of operation in which an HRT around 6 h was applied. In period 2, the concentration of raw sewage increased significantly due to recirculation of sewage sludge from the secondary settlers of the sewage treatment plant. The composition of this "concentrated" sewage, together with a description of "normal" raw sewage and settled sewage during one full year was presented and discussed in chapter 2. Sludge recirculation continued until the end of the experiments (periods 2 to 4). In period 3 (one month), HRT was decreased in the reactor to about 4 h, aiming at a V_{up} of about 0.67 m/h. This V_{up} is expected in 4-m tall UASB reactors when an HRT of 6 h is applied, as recommended by Haskoning and WAU (1994) for tropical regions. This period was intended to force the washout of light particles accumulated in the sludge during its previous operation at a lower V_{up}. At the end of period 3, it was assumed that the washout was completed under the new hydraulic conditions. Period 3 was a preparation for period 4, when the reactor height was increased to 3.95 m, and a V_{up} of about 0.7 m/h was applied during 2 months. The system was finally shut down in October 2002 after more than seven years of operation.

Sewage temperature

The temperature of the influent (settled sewage) and the effluent were continuously monitored with a thermograph Novasen 3752-5-S-C (temperature range: 0-50ºC). Influent temperature was measured with a probe submerged in settled sewage in the distribution tank located after the primary settler, and effluent temperature was measured with a probe installed in the reactor, close to the effluent exit. The thermograph produced a continuous record on circular chart papers with two disposable fiber tip pens (colors red and blue). The chart papers were replaced once a week. Hourly temperature readings were visually extracted from this continuous record and manually loaded on a computer worksheet. Daily means were calculated out of hourly readings. Parallel measurements were performed twice a week in grab samples of raw sewage, settled sewage (influent), and effluent with a digital thermometer (Keithley).

Sample Analyses

Composite samples of raw sewage and the effluents from the settler and the UASB reactor were taken two to three times a week (0.5 L every 3 h for 24 h). Samples were kept at 4ºC until analyzed. Total COD (COD_{tot}), paper-filtered COD (COD_{filt}) (Schleicher & Schuell 595½ 4.4-µm paper filters), and membrane-filtered (dissolved) COD (COD_{dis}) (Schleicher & Schuell ME 25 0.45-µm membrane) were determined in the samples. Suspended and colloidal COD (COD_{sus} and COD_{col}) were calculated as (COD_{tot} - COD_{filt}) and (COD_{filt} – COD_{dis}), respectively. Sludge samples from the reactor were obtained from sampling ports 2, 5, 8, 10, and 12, at 0.19, 0.64, 1.09, 1.39, and 1.69 m from the bottom of the reactor. TSS and VSS were determined in the residue retained by Schleicher & Schuell 589 4.4-µm ashless paper filters. TSS and VSS in the sludge were determined in centrifuged samples, including solids form the supernatant that could be retained by Schleicher & Schuell 589 4.4-µm ashless paper filters. The proportion of granular sludge was determined by filtering a given volume of sludge solution through 3 metallic sieves (pore sizes: 1, 2, and 4 mm). Particles retained in the sieves were considered granules. Analyses were performed according to

APHA *et al.* (1995) or using HACH® micromethods. Statistical comparisons and confidence intervals (CIs) were built at a level of significance (α) of 0.05 (5%).

SMA tests

SMA of the inocula and of the sludge was determined in duplicate 1-L serum bottles. Blanks without substrate were run in parallel to subtract biogas produced by the self-degradation of the sludge (van Lier, 1995; DET, 1994). Up to three feedings of substrate were sometimes provided to the bottles (first feeding: sodium acetate; subsequent feedings: acetic acid). However, SMA was generally calculated with data from the first feeding in order to minimize the effect of bacterial growth on the result. Growth was neglected because the yield coefficient of anaerobic bacteria is only about 5% (Batstone *et al.*, 2002). Therefore, if 1.5 gCOD/L is provided as acetate, assuming 80% conversion, only about 0.06 gCOD will be converted to new bacteria. The SMA calculated after the first feeding would be the closest to the "real" value, provided enough methanogenic bacteria were present at the beginning of the test. Results obtained with second and third feedings were generally similar to those obtained in the first feeding. However, it was also observed that activity decreased along the feedings in some tests, probably due to a toxic effect of the high acetate concentration (much higher than the actual concentration of soluble substrate in the reactor) or decay of specific bacterial populations. Tests were performed at 20 and 30ºC in a refrigerator equipped with an Incutrol® incubator, and a temperature-controlled room, respectively. Methanogenic activity (represented always as SMA) was calculated as follows:

$$SMA = \frac{G \times CF}{VSS \times W \times T}$$

where: SMA = specific methanogenic activity (gCOD-CH_4/gVSS.d); G = methane produced (mL); CF = conversion factor to transform mL-CH_4 at working temperature and pressure conditions to gCOD-CH_4 at a standard temperature of 0ºC and a standard pressure of 1 atm (CF = 0.00223 and 0.00231 gCOD/mL-CH_4 at 30 and 20°C, respectively, assuming an atmospheric pressure at the site of 0.866 atm); VSS = volatile suspended solids concentration in the sludge (gVSS/g of sludge); W = sludge added to the bottle (g); and T = time interval (d). Sludge concentration in the bottles was about 2 gVSS/L.

Effect of V_{up} on COD removal efficiency and sludge characteristics in the UASB reactor

The effect of V_{up} on COD_{tot} removal efficiency was assessed in the UASB reactor during one year within period 1 in order to detect the optimum hydraulic parameters under local conditions. The reactor was operated at different V_{up}s for certain periods of time (or phases) (see **Table 4**, columns 1 to 4). COD_{tot} removal efficiency was measured in composite samples when hydraulic steady state conditions were reached. The criteria for hydraulic steady state were the following: (a) an operation period of more than 10 times the HRT (and more than 2 weeks) (Noyola *et al.*, 1988); and (b) variations in effluent concentration lower than ± 10% (Polprasert *et al.*, 1992). Elmitwalli (2000) and Mahmoud (2002) also considered these criteria satisfactory. A real steady state would only be achieved in the sludge bed, and consequently in the reactor, if the operation period is at least three SRTs (van Haandel and Lettinga, 1994). As it will be shown later, the reactor was in steady state at the beginning of period 1. SMA and VSS concentration of the sludge were measured at different times during and after each phase to assess the effect of these hydraulic changes on the characteristics of the sludge. The SRT in the UASB reactor was calculated as the total amount of sludge (kgVSS) divided by the sludge production (kgVSS/d). The sludge production included the daily amount of VSS retained in the reactor (which should be eventually discharged), and the daily amount of settleable VSS lost with the effluent. The VSS content in the reactor, in turn, was calculated with the average VSS concentration measured in samples obtained along the reactor height and the sludge bed volume at that moment. It was assumed that the sludge composition between sampling ports was homogeneous. Calculations were made according to van Haandel and Lettinga (1994).

RESULTS AND DISCUSSION

Sewage temperature

A summary of more than two years of temperature measurements in reactor influent (settled sewage) and effluent is presented in **Table 1**. Mean monthly temperatures below 20ºC were observed in settled sewage for up to 4 months in a row (see chapter 2). Hourly readings well below the absolute daily minimum were also registered. The difference between reactor influent and effluent temperature was 1.2 ± 0.2ºC (n = 600). This difference was calculated with data from days in which both influent and effluent temperature were measured. This difference is expected to be lower in full-scale plants due to the larger volumes involved.

Table 1. Influent and effluent reactor temperature during more than two years of continuous measurements.

Sample	n	Mean ± CI		Minimum ± CI		Maximum ± CI	
		Day[a]	Month[b]	Day[c]	Month[d]	Day[e]	Month[f]
Settled sewage	613	23.0 ± 0.3	23.1 ± 1.4	12.6 ± 2.4	17.2 ± 0.7	30.0 ± 2.5	26.6 ± 0.7
Effluent UASB	836	22.1 ± 0.2	22.1 ± 1.0	11.0 ± 0.1	14.2 ± 1.2	29.9 ± 0.1	27.0 ± 0.6

[a]Average of daily means calculated out of hourly readings extracted from a continuous record; [b]Average of monthly means calculated out of daily means; [c]Absolute minimum daily mean; [d]Absolute minimum monthly mean; [e]Absolute maximum daily mean; [f]Absolute maximum monthly mean.

Startup and operation

The UASB reactor was started up in August 1995. It was the first experience ever conducted in the country on anaerobic sewage treatment. The criteria for hydraulic steady state were met in about 4 months. At that moment, COD_{tot} removal efficiency reached a constant value of about 40%. This value increased slowly in the following months. Mean HRT during startup was 9.1 ± 0.5 h (V_{up} = 0.28 ± 0.01 m/h). Mean COD_{tot} removal efficiency in the UASB during period 0 was higher than 50%. **Table 2** shows temperature, hydraulic conditions, and removal efficiencies in the system in periods 1 to 4 (results from period 0 are not shown). During the whole experiment, the influent of the UASB reactor can be considered as a very low-strength domestic sewage. In spite of that, COD removal in the reactor was relatively high, and the final effluent concentration was extremely low, in compliance with municipal, provincial, and even European discharge standards for COD_{tot} (European Union Council of Ministers, 1991; Municipalidad de Salta, 2000; SeMADeS, 2001). COD removals observed in the system during period 1 were similar to those obtained by Wang (1994) with anaerobic upflow reactors in series, except for a slightly higher COD_{col} removal in our system.

Table 2. Temperature, hydraulic conditions, COD concentration, and removal efficiencies in system for different periods[a].

Period	Days	Step	Temperature[b] (°C)	Hydraulic conditions		COD	Concentration (mg/L)			Removal efficiency (%)[c]		
				HRT	V_{up} (m/h)		Raw sewage	Effluent Settler	Effluent UASB	Settler	UASB	Total
1	1482-2298	Settler	21,7 ± 0.4	2.0		COD_{tot}	220,2 ± 8,5	151,9 ± 5,3	71,7 ± 3,9	31,0	52,8	67,5
	(816 d)	UASB	22,0 ± 0.7	6.4 ± 0.4	0.40 ± 0.02 h	COD_{sus}	110,7 ± 10,9	74,1 ± 8,1	20,6 ± 2,8	33,1	72,1	81,3
						COD_{col}	29,2 ± 5,6	22,2 ± 3,4	12,3 ± 2,4	23,9	44,3	57,7
						COD_{dis}	62,4 ± 4,4	54,8 ± 4,7	32,8 ± 3,8	12,2	40,1	47,4
2	2298-2541	Settler	22,5 ± 0.4	2.0		COD_{tot}	427,2 ± 43,6	142,7 ± 7,6	72,8 ± 4,6	66,6	49,0	83,0
	(243 d)	UASB	22,7 ± 0.7	6.3 ± 0.3	0.41 ± 0.02 h	COD_{sus}	301,5 ± 35,0	62,7 ± 6,3	23,1 ± 4,6	79,2	63,1	92,3
						COD_{col}	27,6 ± 9,1	20,5 ± 5,5	7,7 ± 6,1	25,5	62,3	71,9
						COD_{dis}	52,1 ± 7,3	51,5 ± 7,1	39,3 ± 6,5	1,2	23,6	24,6
3	2541-2571	Settler	18,9 ± 0.4	2.0		COD_{tot}	378,6 ± 106,7	146,9 ± 18,9	66,4 ± 14,0	61,2	54,8	82,4
	(30 d)	UASB	20,2 ± 0.9	4.1 ± 0.3	0.63 ± 0.04 h	COD_{sus}	304,4 ± 178,1	61,2 ± 13,9	17,7 ± 13,1	79,9	71,1	94,2
						COD_{col}	48,9 ± 27,2	29,0 ± 11,1	16,6 ± 7,0	40,7	42,8	66,1
						COD_{dis}	30,5 ± 9,1	42,6 ± 21,7	24,1 ± 10,5	-39,7	43,4	21,0
4	2571-2623	Settler	21,0 ± 0.4	2.0		COD_{tot}	492,2 ± 87,1	143,5 ± 21,3	78,5 ± 12,9	70,8	45,3	84,0
	(52 d)	UASB[d]	22,9 ± 1.0	5.6 ± 0.1	0.71 ± 0.01 h	COD_{sus}	434,3 ± 155,7	78,8 ± 42,6	34,8 ± 7,7	81,8	55,8	92,0
						COD_{col}	60,5 ± 30,7	36,2 ± 12,2	23,3 ± 21,8	40,2	35,5	61,4
						COD_{dis}	65,3 ± 5,6	66,0 ± 37,1	37,3 ± 26,1	-1,0	43,4	42,9

[a]Mean values ± CIs are provided. [b]Temperature measured in the influent to each unit (grab samples). [c]Removal efficiencies were calculated from average concentrations in each period. [d]Reactor dimensions changed in this period (height = 3.95 m; volume = 0.776 m^3).

Secondary solids recirculated into raw sewage in periods 2 to 4 were completely retained in the settler, which duplicated its removal efficiency, maintaining a rather constant effluent concentration. In period 4, slightly lower removal efficiencies were observed in the UASB reactor due to the increased V_{up}. However, differences in removal efficiency and effluent concentration between the periods were not statistically significant.

Elmitwalli (2000) reported slightly higher removal efficiencies for all COD fractions, except COD_{col}, when treating settled sewage at 13ºC in a UASB reactor, probably because in his study influent concentration was 120% higher, V_{up} was 85% lower (with sludge-wastewater contact facilitated by the small diameter of the reactor), and the reactor had been inoculated with highly active granular sludge. In our study, COD_{dis} represented about 50% of COD_{tot} in the final UASB effluent, in agreement with findings by Halalsheh (2002). Wang (1994) found that 46% of effluent COD_{tot} after anaerobic sewage treatment could be attributed to non-acidified COD_{dis}. As proposed by van der Last and Lettinga (1992), a limited acidification of soluble COD may reduce the maximum possible removal efficiency for anaerobic treatment of (settled) sewage at low temperatures. Remnant volatile fatty acids (VFA) in the final effluent were always about 20 mg/L, an amount that could well account for most of the measured COD_{dis}, depending on its composition. Soluble microbial products (SMP), refractory to anaerobic degradation, could also be responsible for part of the effluent COD_{dis} (Barker and Stuckey, 1999), as suggested by Elmitwalli (2000) and Mahmoud (2002). Halalsheh (2002) reported that 81% of the COD_{dis} in the effluent of a UASB reactor was not anaerobically biodegradable.

TSS and VSS removal efficiencies in the UASB reactor were always high, in spite of low influent concentration (see TSS and VSS concentration in settled sewage in chapter 2). During period 4, TSS and VSS removal efficiencies in the UASB reactor were 77.2 ± 9.0 and 74.0 ± 8.2%, respectively.

Sludge and methane

Methanogenic capacity calculated on the basis of SMA, sludge concentration, and a sludge bed height of only 1 m (assuming that the reactor will always be operated with a higher sludge bed height) was higher than the overall incoming organic loading rate (OLR) (**Table 3**). Therefore, there was always an excess methanogenic capacity in the reactor, even assuming that sewage was completely biodegradable and organic matter was immediately available for methanogenesis. The actual methanogenic activity in the reactor was 0.01 gCOD-CH_4/gVSS.d. The calculation was based on (a) data in **Table 3**; (b) a mean ambient temperature of 23.3°C; (c) a COD_{tot} removal efficiency of about 50%; and (d) a sludge bed height of around 1 m. This calculated value was about 10 and 7 times lower than the SMA at 30 and 20°C, respectively (results at 20°C not shown). In the reactor, the limiting factor is the substrate concentration, while in the lab, an excess of acetate is provided as substrate. A summary of other average parameters in the reactor during one year of operation (periods 2 to 4) is also shown in **Table 3**. SRT was very long, similar to that reported by Elmitwalli (2000). Assuming that SRT was constant since the start up, it can be considered that the reactor was in steady state at the beginning of period 1, after about 1500 d of operation (more than 3 SRTs). The calculated SRT was more than enough to achieve sufficient hydrolysis and methanogenesis at working temperatures (Haskoning and WAU, 1994; Cavalcanti, 2003). Observed SRT was close to the value calculated in chapter 2 with the equation proposed by Zeeman and Lettinga (1999) for the treatment of settled sewage in UASB reactors under local conditions at an HRT of 6 h (V_{up} = 0.67 m/h). Sludge production in the UASB reactor measured during periods 2 to 4 was 2.7% of the reactor volume per month (it represented an increase of about 7 cm/month). Sludge production measured in periods 2 to 4 was higher than expected from previous observations. In fact, the sludge bed was less than 2 m tall at the end of period 1, around operational day 2000. When calculated from the values in **Table 3**, the sludge should have been more than 4 m high in those first 2000 d of operation. It is possible that some sludge was accidentally washed out without notice during previous periods. No intentional sludge discharges were performed during the operation of the reactor, other than those described in chapter 4. In any case, the sludge bed height was always within the optimum range for a safe operation (see chapter 4).

Table 3. Average parameters in the UASB reactor during one year of operation (periods 2 to 4).

Parameter	Value
OLR ($kgCOD_{tot}/m^3_{reactor}.d$)	0.60
SMA ($gCOD\text{-}CH_4/gVSS.d$)	0,098 ± 0.016
Sludge concentration (gVSS/L of sludge)	28.6 ± 4.0
Reactor excess methanogenic capacity (%) (assuming a sludge bed height of 1 m)	122
SRT (d)	498
Sludge growth rate (cm/d)	0.23
Sludge production ($kgCOD/kgCOD_{removed}$)	0.18
Methane gas recovery ($Nm^3/kgCOD_{removed}$)	0.14
Methane content in the biogas (%)	95

About 30% of the sludge in the UASB reactor was bigger than 1 mm in diameter and was considered granular. Big granules (some up to 5 cm in diameter) were observed, probably formed from aggregation of smaller ones (**Figure 2**). The biggest granules showed a tendency to float, and were apparently weaker than smaller ones. The development of granules can be explained by a low concentration of suspended solids and COD in the influent (Hulshoff Pol, 1989), and an adequate combination of HRT and V_{up} (Alphenaar *et al.*, 1993).

Figure 2. Granules from the UASB reactor.

Average methane production in periods 2 to 4 was slightly lower than values reported for raw sewage by Lettinga *et al.* (1983) (0.21 $Nm^3/kgCOD_{removed}$), Noyola *et al.* (1989) (0.17 $Nm^3/kgCOD_{removed}$), and that expected in full-scale UASB reactors (Haskoning and WAU, 1994). This can be explained by the fact that a considerable proportion of COD_{sus}, the most biodegradable fraction, was already retained in the settler. Methane content in the biogas was very high (90-95%) and constant throughout the experiments.

A COD balance was attempted over the reactor, but confusing results were obtained. In fact, when dissolved methane (calculated according to Henry's law) was included in the balance, the COD going out of the reactor was greater than the COD entering the reactor with the influent. Possible explanations are the following:

(a) Despite the fact that some gaseous methane was recovered, the assumption made to use Henry's law, namely that the effluent was completely saturated with methane, might not hold true because the biogas production was very small.
(b) The quantification of the exact amount of sludge in the reactor is subject to some errors because the sampling ports were separated 15 cm from each other. This error can be particularly sensitive when the sludge growth rate is very low.
(c) The extraction of representative sludge samples to determine COD was difficult, especially because the sludge was not homogeneous along the reactor height.
(d) The influent concentration was always very low and therefore, errors in the balance can be magnified.

Results from the COD balance are not shown, although it can be calculated with the information provided in this chapter. A detailed methodology for the construction of COD balances is presented in chapter 5, together with actual examples of balances over UASB reactors for sewage treatment.

SMA and VSS profiles in the anaerobic sludge

SMA and VSS concentration varied considerably along the sludge bed height. The sludge was more diluted at the bottom (port 2) and at the top of the reactor (port 12). The highest VSS concentration and the larger proportion of big granules was observed in the middle sections of the reactor (ports 5 and 8). SMA showed a trend to decrease when VSS concentration increased (**Figure 3**).

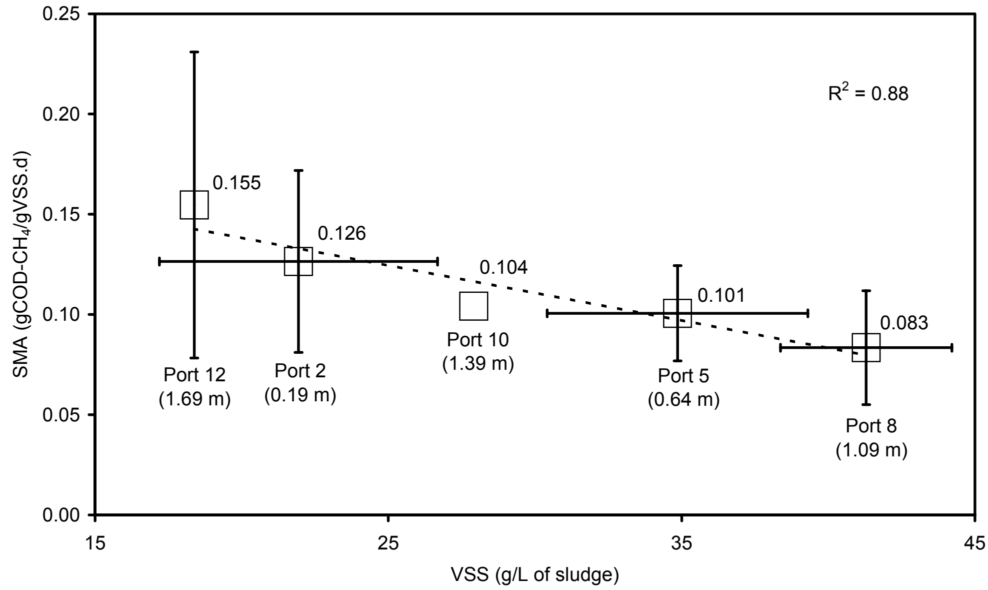

Figure 3. SMA against VSS concentration in the sludge of the UASB reactor. Linear regression is shown as dotted line. CIs are shown as error bars.

Probable causes of this trend might be internal mass transfer limitations in big granules, the accumulation of inert organic material in the core of the granules, and the clogging of internal channels by cell lysis and bacterial growth, as proposed by Alphenaar (1994). High variability was observed between measurements at different times in ports 2 and 12 (bottom and top of the sludge bed). Representative sampling was difficult from these two ports due to disturbances produced by raising biogas bubbles (port 12) and by localized mixing induced by the influent distribution pipe (port 2). Sludge stratification was also reported in upflow sludge bed and filter (UBF) reactors fed with a concentrated synthetic sugar wastewater (Guiot *et al.*, 1992), in UASB reactors fed with a mixture of glucose and potassium acetate (Kalyuzhnyi *et al.*, 1996), in anaerobic sludge from fluidized bed reactors treating acetate as a sole carbon source (Hidalgo and García-Encina, 2002), and in UASB reactors treating raw sewage (Leitão, 2003). The ratio VSS/TSS was approximately 0.5 in all measurements at different heights, an indication that the sludge was well stabilized. Results on the stability of this sludge will be presented and discussed in chapter 5.

Effect of V_{up} on COD removal efficiency and sludge characteristics in the UASB reactor

In this experiment, different hydraulic conditions were imposed to the UASB reactor during five different phases (see first 4 columns in **Table 4**). Phases are indicated with circles in **Figure 4** against the background of conditions applied during the entire operation of the reactor (2622 d; 7.18 y). Data points represent days in which influent flow rate was checked on the site. It can be assumed that flow rate was constant between data points. This experiment began 1482 d (more than 4 y) after the reactor was started up and lasted 354 d (indicated with the horizontal bar in **Figure 4**). During the experiment, the sludge never exceeded port 8 (109 cm from the bottom of the reactor; 42.7% of the reactor height).

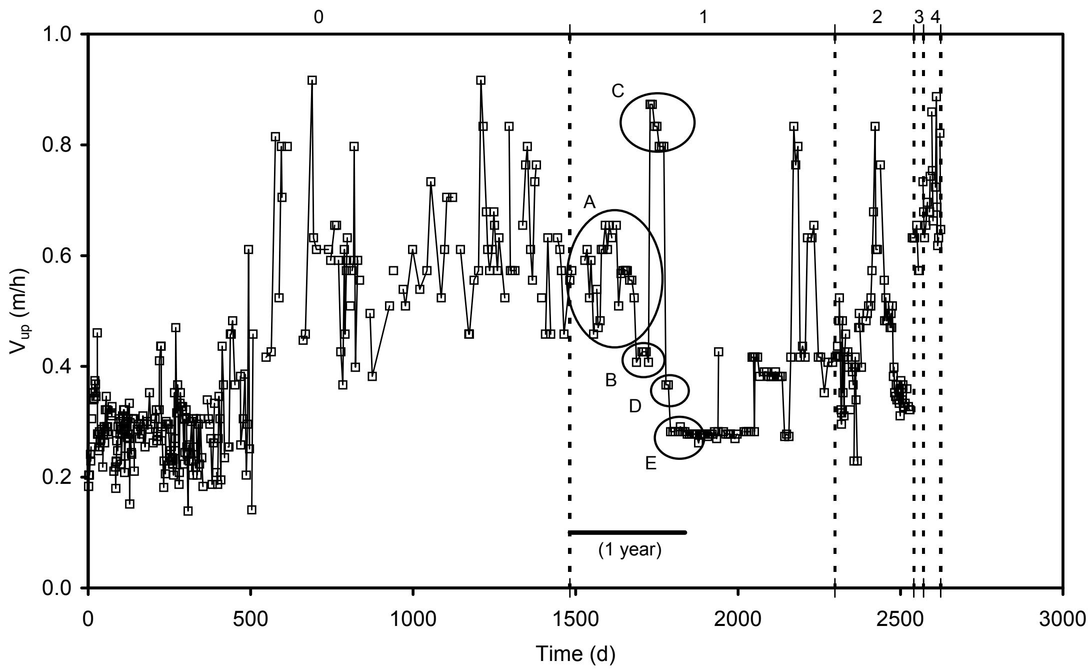

Figure 4. V_{up} against time for the entire operation of the UASB reactor. Circles indicate hydraulic changes imposed during the study of the effect of V_{up} on COD_{tot} removal efficiency. The horizontal bar shows the duration of this experiment. Numbers refer to operational periods. Letters refer to experimental phases.

COD removal efficiency

The duration of hydraulic changes applied during the different phases represented only a minor fraction of the time needed to reach a new steady state in the sludge (i.e. a one month-long operational change would represent only 2% of that time, based on an SRT of 500 d). For that reason, sludge characteristics were assumed to be constant in spite of hydraulic changes and therefore, variations in COD removal efficiency reflected physical phenomena rather than variations in the treatment capacity of the sludge. The results are presented in **Table 4** and **Figure 5**. COD_{tot} removal efficiency observed in phase B (V_{up} = 0.42 m/h; HRT = 6.1 h) was significantly higher than in all other phases, with the exception of phase D (V_{up} = 0.37 m/h; HRT = 7.0 h).

Table 4. COD_{tot} removal efficiency in the UASB reactor at different hydraulic conditions.

Phase	Duration (d)	HRT (h)	V_{up} (m/h)	COD_{tot}	
				Influent (mg/L)	Removal (%)
A	198	4.5 ± 0.2	0.57 ± 0.02	164.5 ± 12.7	52.2 ± 3.3
B	37	6.1 ± 0.1	0.42 ± 0.01	131.1 ± 13.5	63.2 ± 6.1
C	42	3.1 ± 0.1	0.83 ± 0.02	138.1 ± 11.2	46.1 ± 6.3
D	7	7.0 ± NA[a]	0.37 ± NA[a]	150.4 ± 19.7	59.2 ± 5.1
E	44	9.0 ± 0.1	0.28 ± 0.003	169.6 ± 20.5	53.4 ± 5.3

[a]Not applicable (only two values).

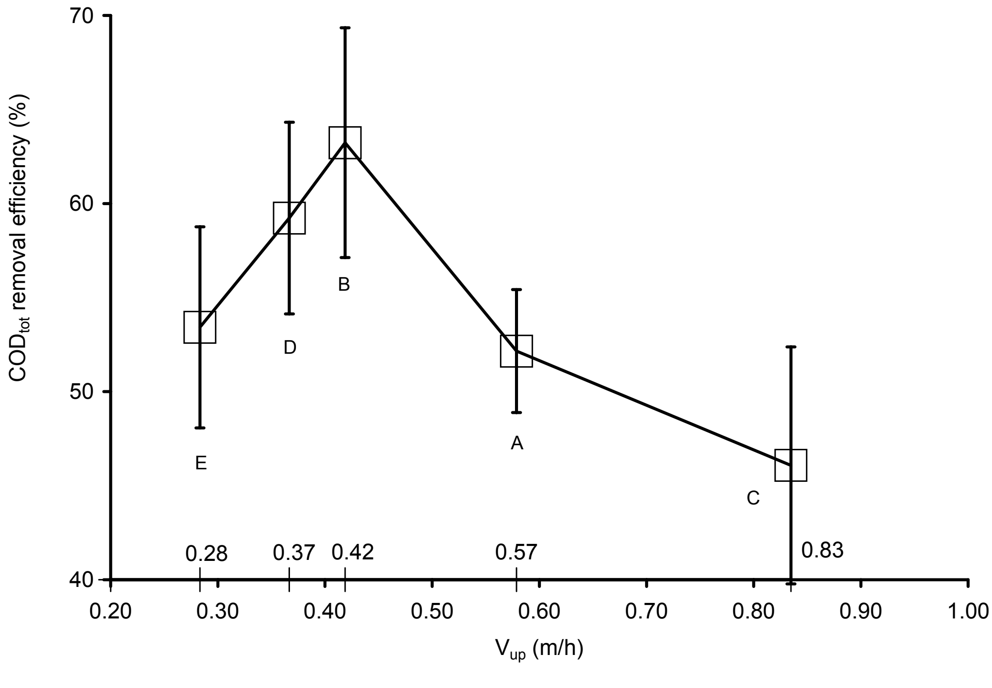

Figure 5. COD_{tot} removal efficiency as a function of V_{up}. This figure was built with data from columns 4 and 6 in **Table 4**. Letters refer to experimental phases. CIs are shown as error bars.

Influent concentration was assumed to be relatively constant between phases despite some variations observed (**Table 4**). In any case, the maximum removal efficiency was measured with the lowest influent concentration (phase B), reinforcing the idea that there is a quantifiable effect of the

V_{up} on the COD_{tot} removal efficiency. The poorer retention of TSS and VSS (and consequently suspended COD) may explain lower removal efficiencies at high V_{up}. On the other hand, the drop observed below 0.42 m/h could be ascribed to insufficient sludge-wastewater contact caused by channels in the sludge bed and/or lower biogas production. In fact, OLR is lower at lower V_{up} (with a relatively constant influent concentration) and therefore biogas production will also be lower.

The effect of V_{up} and HRT on the removal of all COD fractions could give more insight in the processes involved. For practical reasons, V_{up} was first decreased in phase B, then increased in phase C, and then decreased again in phases D and E (see **Table 4** and **Figure 4**). However, as hydraulic steady state conditions were reached in all cases (except probably for the short phase D), the way in which hydraulic changes were applied was considered irrelevant and should not affect the results.

Sludge SMA and VSS concentration

No significant differences in SMA and VSS have been detected in the sludge as a whole as seen in samples taken during and after the different phases (samples were not taken in phase D) (**Figure 6**).

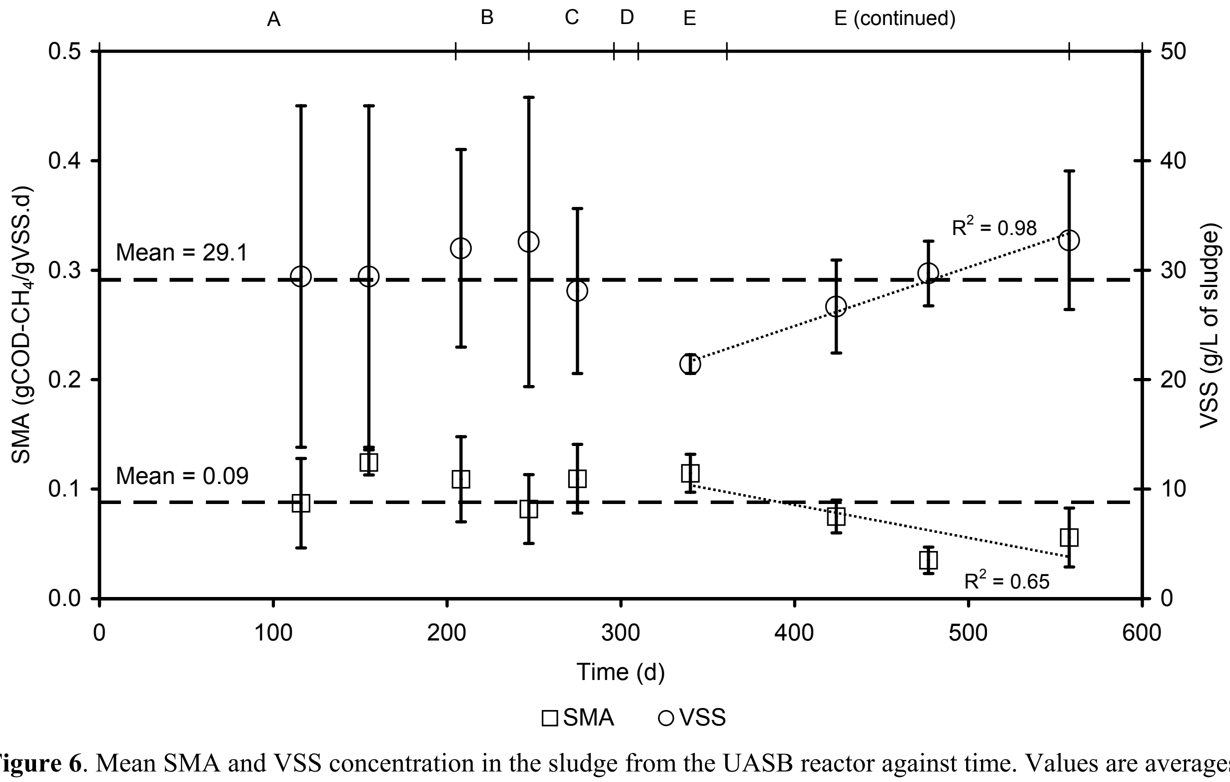

Figure 6. Mean SMA and VSS concentration in the sludge from the UASB reactor against time. Values are averages over the entire sludge bed. Phases are indicated above the chart. Average values are indicated with dashed lines. Linear regressions are shown as dotted lines. Error bars represent CIs.

This observation endorsed the assumption that sludge characteristics were constant during considerable periods of time, irrespective of changes in hydraulic conditions. No correlation whatsoever was observed between SMA and V_{up}. On the contrary, other studies showed that, when hydraulic conditions were imposed to the reactor until a new steady state was reached, a positive effect of V_{up} on the SMA was observed on anaerobic sludge grown either on soluble synthetic substrates (Guiot *et al.*, 1992; O'Flaherty *et al.*, 1997) or raw sewage (Leitão, 2003). In this study, VSS increased steadily when a low V_{up} of 0.28 m/h was applied for a relatively long time (see last 4 data points in **Figure 6**), probably due to the accumulation of undegraded VSS originated in poor sludge expansion at that low V_{up}. In the same period, SMA showed a slight trend to decrease accordingly (although it increased somewhat in the end). It is likely that a new steady state was being reached in the sludge bed under these constant hydraulic conditions, with its unique combination of SMA and VSS. Unfortunately, there are no comparable SMA and VSS measurements from the first 500 d of reactor operation, in which a similar V_{up} was applied (see **Figure 4**). In **Figure 6**, the high variability within the sludge bed in both SMA and VSS concentration is reflected in the CIs. SMA measurements at lower (or ambient) temperatures could be useful to determine the behavior of the sludge bed under local conditions, and for design purposes (i.e. to estimate biogas production in full-scale reactors). Average SMA (at 30°C) during this experiment was 0.088 ± 0.019 gCOD-CH_4/gVSS.d, with 29.1 ± 2.3 gVSS/L of sludge (dashed lines in **Figure 6**). Averages from 18 tests performed along almost three years (from day 1598 to day 2647) were presented in **Table 3** (detailed results not shown).

CONCLUSIONS

- A system consisting of a settler followed by a UASB reactor efficiently treated domestic sewage under subtropical conditions.
- The final effluent concentration was always extremely low, in compliance with discharge standards of 125 mgCOD_{tot}/L.
- Removal efficiencies up to 84.0% in total COD and 92.0% in suspended COD have been observed in the entire system at a mean HRT of 2 h in the settler and 5.6 h in the reactor (V_{up} = 0.71 m/h). Maximum COD_{tot} removal efficiency in the reactor (63.2 %) was observed at a V_{up} of 0.42 m/h (HRT = 6.1 h).
- A granular sludge bed developed in the UASB reactor, which had been inoculated with semi-digested sewage sludge. Granules up to 5 cm in diameter were observed, apparently formed from aggregation of smaller ones.
- The SRT in steady state conditions was 498 d. The sludge growth rate was 2.7% of the reactor volume per month. Sludge characteristics (VSS concentration and SMA) were relatively constant despite variations applied in flow rate. Mean SMA in the anaerobic sludge was 0.10 gCOD-CH_4/gVSS.d. Sludge concentration was 28.6 gVSS/L of sludge.

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CHAPTER 4

The effect of sludge discharges and upflow velocity on the removal of suspended solids in an upflow anaerobic sludge bed (UASB) reactor treating settled sewage under subtropical conditions

ABSTRACT

The removal of total and volatile suspended solids (TSS/VSS) was studied in a pilot-scale upflow anaerobic sludge bed (UASB) reactor treating settled sewage under subtropical conditions in Salta, Argentina. The effect of the sludge bed height (h_{sb}) and the upflow velocity (V_{up}) on the removal of TSS and VSS was assessed. TSS and VSS removal efficiencies higher than 90% have been achieved when V_{up} was 0.43 m/h or lower (equivalent to a hydraulic retention time of 6 h or higher) and, at the same time, h_{sb} was 0.92 m or higher. Effluent concentration was always extremely low (lower than 10 mgTSS/L). TSS and VSS removal efficiencies were inversely proportional to the V_{up}. The reactor was operated at 0.85 m/h during periods of up to two days, but no significant sludge washout was observed, even after heavy sludge discharges were performed. A safe and efficient operation could be achieved in 4-m tall UASB reactors treating settled sewage keeping h_{sb} between 1 and 2 m, and applying a V_{up} around 0.5 m/h. The specific methanogenic activity (SMA) of the sludge did not change significantly after discharges. As the sludge growth rate was very low (less than 3% of the reactor volume per month), one discharge every two years could be enough to dispose of the excess sludge. TSS/VSS removal efficiency could be a useful criterion to decide on the right moment for sludge discharges.

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INTRODUCTION

The removal of suspended solids (SS) is one of the main objectives of sewage treatment. The presence of high concentrations of SS in the influent, the slow degradation of SS entrapped in the sludge bed, and the washout of incoming SS and/or biological sludge are cited as the main causes of bad effluent quality in upflow anaerobic sludge bed (UASB) reactors treating sewage below 20ºC (Elmitwalli, 2000). Pre-settling of sewage in primary settlers, two-stage anaerobic systems, and hybrid reactors have been proposed to improve the retention and degradation of suspended solids under these conditions (van Haandel and Lettinga, 1994; Wang, 1994; Elmitwalli, 2000). In primary settlers, total COD removal efficiencies higher than 50% can be achieved, with total and volatile SS (TSS/VSS) removal above this value (Metcalf and Eddy, 1991). However, no matter how efficient the settler might be, there will always be SS to retain in subsequent processes. Besides, SS in the effluent of a settler are, by definition, more difficult to retain than those removed. UASB reactors are very efficient at retaining SS from sewage, especially in tropical regions (van Haandel and Lettinga, 1994; Cavalcanti, 2003). SS removal in UASB reactors depends on the type of sewage and the combined effect of the sludge bed height (h_{sb}) and the liquid upflow velocity (V_{up}) in the reactor, the latter parameter related to the hydraulic retention time (HRT) and the reactor height. In principle, low h_{sb} and/or high V_{up} would hinder the retention of particles and colloids. In any case, it was reported that UASB reactors are, in general, not very effective at removing colloidal matter, no matter what the hydraulic conditions are (Elmitwalli, 2000). The effect of h_{sb} and V_{up} on the removal of SS needs to be assessed to optimize the design and performance of UASB reactors for the treatment of settled sewage at low temperatures.

One of the advantages of anaerobic treatment over aerobic treatment is the fact that biological sludge production is low (see chapter 1). Nevertheless, some sludge has to be discharged from the system at regular intervals, and this operation should not affect the stability and performance of the process (Cavalcanti *et al.*, 1999). To minimize process disturbances, operational costs, and human errors, sludge discharges should be performed at a minimum frequency, and a maximum of sludge should be discharged each time. When treating a more complex wastewater like raw sewage the excess sludge production is mainly dictated by the concentration of inert SS in the sewage (Janssen *et al.*, 2003). In this case, the differences between aerobic and anaerobic systems become smaller, especially in terms of wet weight of excess sludge. Differences may still be significant in terms of volume, depending on the TSS concentration in anaerobic and aerobic excess sludge.

The objectives of this work were the following: (a) to study TSS and VSS removal efficiency in a pilot-scale UASB reactor treating settled sewage under subtropical conditions; (b) to assess the effect of h_{sb} and V_{up} on TSS/VSS removal efficiency; and (c) to determine the minimum h_{sb}, and the maximum V_{up} that could still guarantee a safe and efficient reactor operation (defined in terms of good effluent quality, and high TSS/VSS removal).

MATERIALS AND METHODS

Experiments were performed in Salta, Argentina, where ambient temperature is 16.5 ± 0.2 ºC (Arias and Bianchi, 1996). See a detailed description of this location in chapter 2. The UASB reactor used in this work was part of the pilot plant described in chapter 3. The reactor was fed with sewage previously submitted to preliminary treatment (screens and grit chamber) and 2 h of settling in a primary sedimentation tank. The UASB reactor (volume = 0.501 m^3; height = 2.55 m; diameter = 0.5 m) was started up in 1995, and was at steady state at the beginning of these experiments, after more than five years of operation. More than 30% of the sludge bed in the reactor was granular. Influent and reactor temperatures were continuously monitored with a thermograph (Novasen 3752-5-S-C).

Effect of h_{sb} and V_{up} on TSS/VSS removal

A test was designed to study the separate and combined effect of the h_{sb} and the V_{up} on the removal of TSS and VSS, in order to obtain values for the design and operation of UASB reactors treating settled sewage under subtropical conditions. TSS and VSS removal efficiency were determined in influent and effluent samples from the reactor under different combinations of h_{sb} and V_{up}. At the beginning of the experiments h_{sb} was 1.84 m. Sludge was discharged from the reactor three times at regular intervals of time. Discharges amounted to 90 L each, equivalent to 25% of the initial volume of sludge in the reactor. After each discharge, TSS and VSS removal efficiencies were assessed at V_{up}s of 0.28, 0.43, and 0.85 m/h (HRT = 9, 6, and 3 h, respectively). Results presented in chapter 3 showed that the TSS and VSS removal efficiency was higher than 70% in the UASB reactor at steady state. Based on these results, a threshold level of 70% removal efficiency was established as the minimum acceptable performance of the reactor. When the combination of h_{sb} and V_{up} applied to the reactor lead to removal efficiencies lower than 70% in either TSS or VSS, this particular combination was considered unsafe or unacceptable.

For the sake of consistency, the first V_{up} assessed after sludge discharges was 0.28 m/h, increasing subsequently to 0.43 and 0.85 m/h. In this way, washout of light particles immediately after sludge discharges was minimized. Each V_{up} was applied for about two days before sampling. Although this time represented always more than three HRTs, it was not enough to reach hydraulic steady state as defined by Noyola *et al.* (1988), namely that of an operation period of more than ten times the HRT (and more than two weeks) (see chapter 3). However, variations in effluent concentration were lower than ± 10% when samples were taken, as recommended by Polprasert *et al.* (1992). The reactor was in steady state at the beginning of these experiments, performed during the so-called “period 1” (see chapter 3). In any case, two days were certainly not enough to reach a new steady state in the sludge bed, where the calculated solids retention time (SRT) was almost 500 days (see also chapter 3). Under these circumstances, variations in TSS/VSS removal efficiency were assumed to be a response to hydraulic conditions and changes in the retention capacity of the sludge bed, reflecting physical phenomena rather than changes in the biological characteristics of the system.

To avoid (organic) overloading of the reactor, the methanogenic capacity (M, in $kgCOD/m^3_{reactor}.d$) of the sludge bed was always kept higher than the biodegradable organic matter (B, in $kgCOD/m^3_{reactor}.d$) retained in the reactor, even when assuming that all the particulate organic pollutants were hydrolyzed at once and became immediately available for methanogenesis (**Table 1**). To detect possible organic overloading, the concentration of volatile fatty acids (VFA) was also monitored in the effluent. Terms M and B in **Table 1** were calculated with equations 1 and 2.

$$M = \frac{MA * VSS * V_s}{V_r} \tag{1}$$

$$B = OLR * ABD * RE \tag{2}$$

where MA = methanogenic activity at a working temperature of 20ºC (kgCOD-CH_4/kgVSS.d); VSS = volatile suspended solids concentration in the sludge bed (kg/m^3 of sludge); V_s = sludge volume left in the reactor after discharges (m^3); V_r = reactor volume (m^3); OLR = total organic loading rate ($kgCOD_{influent}/m^3_{reactor}.d$); ABD = total anaerobic biodegradability of settled sewage (%); and RE = total COD removal efficiency in the reactor (%). The following parameters were used to calculate M and B:

(a) MA = 0.065 gCOD-CH_4/gVSS.d. This is about 65% of the sludge specific methanogenic activity (SMA) reported in chapter 3 for this particular sludge (in chapter 5 it was reported that methanogenic activity at 20°C was about 65% of the SMA at 30°C).
(b) VSS = 30 g/L of sludge (chapter 3).
(c) V_s = calculated at each h_{sb} (reactor diameter = 0.5 m).
(d) V_r = 0.5 m^3.
(e) OLR = 1.15 $kgCOD/m^3_{reactor}.d$. This value was calculated for an HRT of 3 h (the worst scenario) and a total COD concentration in the influent of 143.3 mg/L (as reported in chapter 2).
(f) ABD of settled sewage at 23ºC = 40%. This value was calculated on the basis of anaerobic biodegradability measurements in raw sewage (chapter 2), assuming that the biodegradability of the different COD fractions was the same in raw and settled sewage. Settled sewage is likely to be less biodegradable than raw sewage due to its lower amount of highly biodegradable suspended solids.
(g) RE = 50% (as reported in chapter 3 for the reactor at steady state at an HRT = 6.3 h and a V_{up} = 0.41 m/h).

Table 1. Conditions in the sludge bed at the beginning of the experiments, and after sludge discharges. V_s = sludge volume left in the reactor after discharges; h_{sb} = sludge bed height after discharges; M = methanogenic capacity after discharges; B = (biodegradable) organics retained in the reactor. Parameters used for the calculation of M and B are presented in the text.

	V_s	h_{sb}	M	B	M/B
	(L)	(m)	($kgCOD/m^3_{reactor}.d$)		(-)
Initial conditions	361.3	1.84	1.41	0.23	6.1
First discharge	271.3	1.38	1.06	0.23	4.6
Second discharge	181.3	0.92	0.71	0.23	3.1
Third discharge	91.3	0.46	0.36	0.23	1.6

Sampling and analyses

Around two days after each sludge discharge and/or change in the operational V_{up}, influent and effluent samples were taken and analyzed for TSS and VSS during one day. Personnel from the sewage treatment plant took composite samples during 24 h (1 L every 3 h) and graduate students from the lab took grab samples in duplicate the same day at 09:20, 15:20 and 23:20 h. Four additional grab samples were withdrew every 30 min immediately after discharges and changes in V_{up}, in order to detect sludge washout events. Samples were kept at 4°C before analyzed. In the lab, samples where stirred for 20 seconds in a magnetic stirrer to ensure homogeneity, and a sub-sample was filtered in a Büchner funnel through Schleicher & Schuell Nº189 ashless paper filter (pore diameter = 4.5 μm). TSS and VSS were determined in the retained solids according to APHA *et al.* (1995).

Statistical analysis

A selected set of results was processed with one-way analysis of variance (ANOVA) (Steel and Torrie, 1980). The "treatments" were either h_{sb} or V_{up}, and the variables assessed were the removal of TSS and VSS. The effect of the h_{sb} on TSS/VSS removal was assessed when the V_{up} was the lowest (0.28 m/h) in order to minimize the collateral effect of the V_{up} on the removal of SS. In fact, higher V_{up}s could have lead to a reduction of the removal of SS, or induced washout of sludge when high amounts of sludge were present in the reactor (high h_{sb}). In such a situation, the effect of the h_{sb} could have been erroneously attributed to the effect of the V_{up}. On the contrary, the effect of V_{up} on TSS/VSS removal was statistically assessed only when h_{sb} was 0.92 and 1.38 m (50 and 75% of the initial sludge bed height, respectively). At these heights, a minimum washout of sludge and a high entrapment of suspended solids were expected. Results obtained at h_{sb} = 1.84 m were not submitted to statistical analysis because some washout of sludge due only to the expansion of the sludge bed was expected when high V_{up}s were applied to the reactor. On the other hand, results obtained when h_{sb} = 0.46 m were neither statistically assessed because in this case, the amount of sludge present in the reactor may have been insufficient to retain suspended solids, and this effect could have been erroneously attributed to the V_{up}. In this way, ANOVA tests were intended to detect the independent effects of both h_{sb} and V_{up} on the removal of TSS and VSS. Unless indicated otherwise, statistical comparisons and confidence intervals were built at a significance level (α) of 0.05 (5%).

RESULTS AND DISCUSSION

Average performance

Influent and reactor temperatures during this experiment are shown in **Table 2**.

Table 2. Influent and reactor temperature (ºC).

Measuring point	Mean ± CI[a]	Monthly averages		Absolute values	
		Minimum	Maximum	Minimum	Maximum
Influent	22.9 ± 2.8	17.0	26.6	9.9	29.9
Reactor	22.1 ± 3.7	14.2	27.0	11.5	30.0

[a]CI = confidence interval.

Mean TSS and VSS concentrations in settled sewage (the influent to the reactor) were 37.4 ± 6.5, and 11.1 ± 2.2 mg/L, respectively. Average removal efficiencies observed during the whole experimental period were 80.3 ± 8.6% (TSS) and 75.9 ± 9.7% (VSS) for grab samples, and 86.5 ± 4.9% (TSS) and 77.0 ± 8.5% (VSS) for composite samples. In spite of the low influent concentration, average removal efficiencies were high, irrespective of different conditions applied. Consistently, TSS and VSS measurements in influent and effluent were slightly lower in composite samples than in grab samples probably because sampling hours were different. Effluent concentration was always lower than 10 mg/L for both TSS and VSS. VFA concentration in the effluent was always lower than in the influent, indicating that methanogenesis was never exceeded (results not shown).

The effect of h_{sb} and V_{up}

The first sludge discharge was performed from a sampling port located 1.09 m from the bottom of the reactor, where SMA was the lowest (from results reported in chapter 3). After the first sludge discharge and up until the end of the experiment, SMA was similar all along the sludge bed (around 0.1 gCOD-CH_4/gVSS.d), and the second and third sludge discharges were then performed from the top section of the sludge bed. **Figure 1** shows mean TSS (left) and VSS (right) removal efficiencies against V_{up} for all h_{sb} studied (based on grab samples). TSS removal efficiency fell below 70% only when h_{sb} = 0.46 m and, at the same time, V_{up} = 0.85 m/h. On the other hand, VSS removal was lower than 70% when h_{sb} = 0.46 m, irrespective of the V_{up} applied. When h_{sb} was 0.92 m or higher, TSS and VSS removal efficiencies were always higher than 70% (with the exception of a VSS removal slightly lower than that when h_{sb} was 1.38 m and V_{up} was 0.85 m/h). Best results were achieved when $h_{sb} \geq 0.92$ m, and $V_{up} \leq 0.43$ m/h (HRT ≥ 6 h), with some TSS/VSS removal efficiencies higher than 90% (indicated with circles in **Figure 1**). The worst combination was V_{up} = 0.85 m/h (HRT = 3 h) and h_{sb} = 0.46 m, which yielded removal efficiencies of only 55.0 ± 13.6 and 35.8 ± 16.5% for TSS and VSS, respectively. De Man *et al.* (1986) reported that a V_{up} exceeding 0.5 m/h resulted in a significant decrease in SS removal in the treatment of domestic sewage in a UASB reactor at low temperatures.

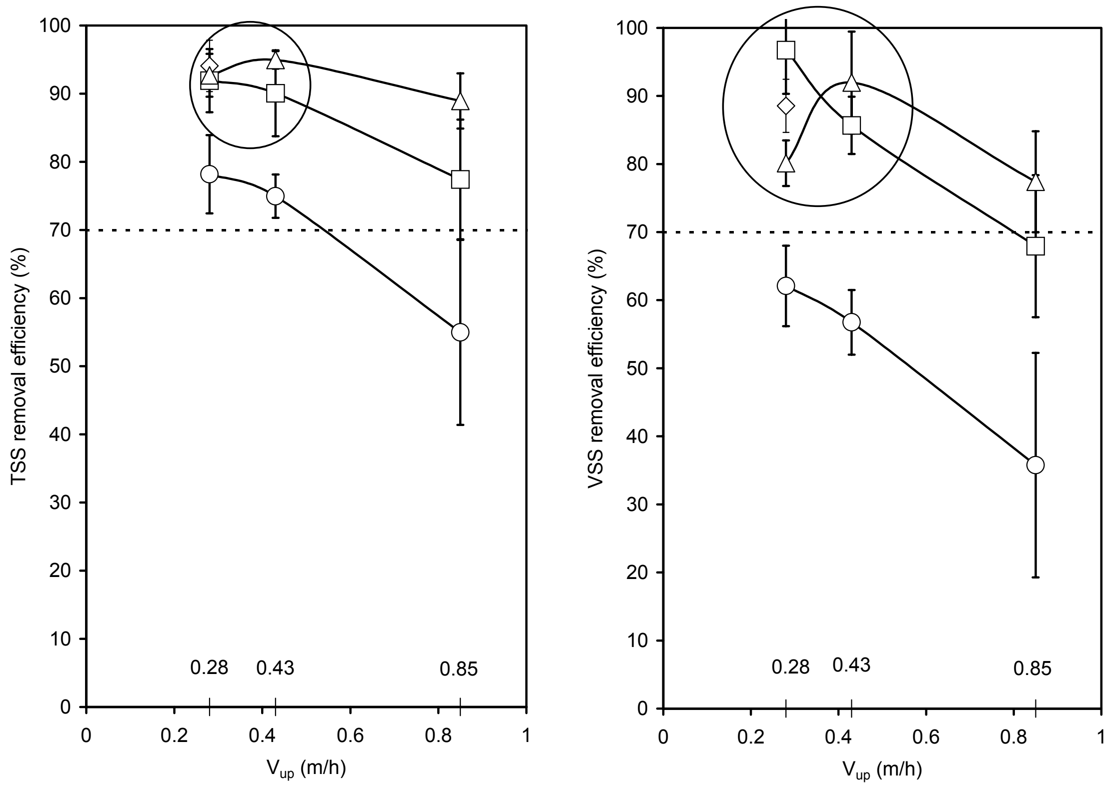

Figure 1. TSS (left) and VSS (right) removal efficiency against V_{up} for different h_{sb} (◊ = 1.84 m; □ = 1.38 m; Δ = 0.92 m; O = 0.46 m). Values shown are averages from three daily grab samples analyzed in duplicate the day after the reactor was in (hydraulic) steady state. Confidence intervals are shown as Y error bars.

From **Figure 1** it seems that VSS were more sensitive than TSS to the combined effect of V_{up} and h_{sb} probably because the lightest particles had a higher proportion of VSS. The reactor was operated at 0.85 m/h during periods of up to two days, but no significant sludge washout was observed, even after up to 50% of the sludge bed was discharged (h_{sb} decreased from 1.84 m to 0.92 m), in agreement with laboratory results reported by Cavalcanti *et al.* (1999). For design purposes, a V_{up} around 0.5 m/h and an h_{sb} higher than 1 m are recommended. When the UASB reactor was at steady state and h_{sb} was around 1.80 m (0.7 m below the effluent exit, located at 2.55 m), TSS and VSS removal efficiencies higher than 70% were observed at a V_{up} of 0.41 m/h (HRT = 6.3 h) (results obtained during the period 2 described in chapter 3). Therefore, it could be said that in 4-m tall reactors, the sludge bed could safely rise up to 3 m (1 m below the effluent exit) without a reduction in the quality of the effluent, as long as there are no sharp flow rate peaks in the influent. More studies are needed to clearly establish the maximum acceptable h_{sb} in taller reactors. Sludge production in the UASB reactor was 0.18 kgCOD/kgCOD$_{removed}$. The sludge COD was 45.0 ± 3.0 g/L of sludge. This sludge production represented about 7.0 cm per month, or 2.7% of the reactor volume per month. At this growth rate, it would take more than 4 years to fill up a 4-m tall UASB reactor. If the maximum allowable h_{sb} is 3 m, and the recommended h_{sb} after discharges is 1 m, it means that sludge discharges can be safely performed once every two years. **Figure 2** shows average values for TSS and VSS removal efficiencies against h_{sb} (left) and V_{up} (right), when their effects were considered independently from each other.

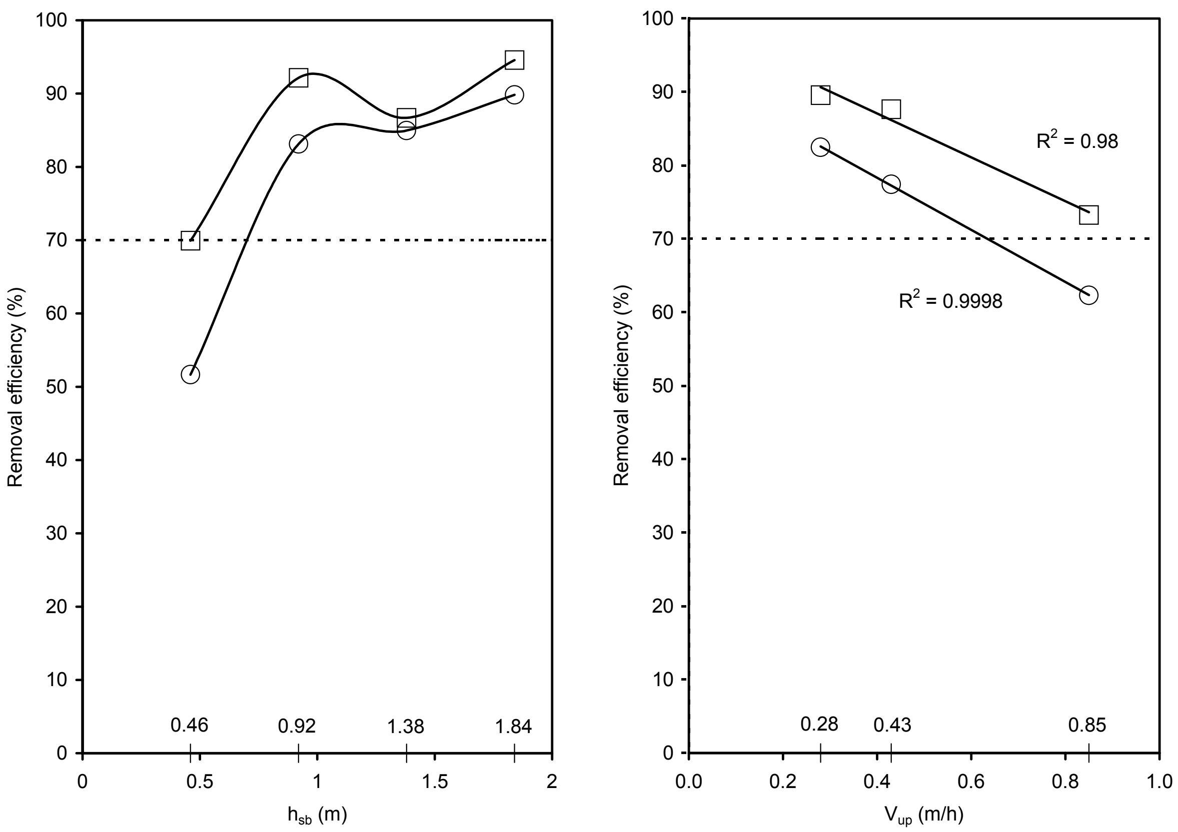

Figure 2. Average TSS (□) and VSS (O) removal efficiency against h_{sb} (left) and V_{up} (right).

Results obtained at each h_{sb} were averaged for all V_{up} applied, and vice versa. TSS/VSS removal efficiencies were always higher than 70% when $h_{sb} \geq 0.92$ m, but dropped abruptly when $h_{sb} = 0.46$ m (**Figure 2**, left). On the other hand, TSS/VSS removal seems to be inversely proportional to V_{up}, judging by the high regression coefficients obtained (**Figure 2**, right). The reactor could deal successfully with simultaneous hydraulic shock loads and heavy sludge discharges, as massive sludge washout and/or negative TSS/VSS removals were never detected at any of the applied conditions. However, a $V_{up} \geq 0.85$ m/h and an $h_{sb} \leq 0.46$ m or lower should be avoided in order to achieve high TSS/VSS removal efficiencies and to guarantee a safe reactor operation.

ANOVA tests

Results obtained in ANOVA tests are summarized in **Table 3**. When $h_{sb} = 1.38$ m, TSS removal at $V_{up} = 0.28$ m/h was significantly higher than at 0.85 m/h ($\alpha = 0.1$), but not significantly different from that at 0.43 m/h. No significant differences were found between 0.43 and 0.85 m/h, which was attributed to the internal variability of the data. At the same h_{sb}, VSS removals at $V_{up} = 0.28$ and 0.43 m/h were significantly higher than at 0.85 m/h ($\alpha = 0.1$). No significant differences were found between 0.28 and 0.43 m/h. TSS and VSS removal efficiencies were not significantly different for all V_{up} applied when $h_{sb} = 0.92$ m. On the other hand, TSS and VSS removal efficiencies were not significantly different for all h_{sb} applied when $V_{up} = 0.28$ m/h. Results from the ANOVA tests also show that the influence of V_{up} on the removal of TSS and VSS was negligible when the sludge bed in the reactor was 0.92 m. Washout of light sludge may explain significant differences observed between different V_{up} when $h_{sb} = 1.38$ m. The effect of h_{sb} was irrelevant when V_{up} was 0.28 m/h. These results confirm that a good combination of both variables is important to guarantee high TSS/VSS removal efficiencies and the lowest possible effluent concentration.

Table 3. Results of the ANOVA tests.

Variables		Relationships between treatments	
Fixed	Treatments	Related to TSS removal (%)	Related to VSS removal (%)
h_{sb} = 1.38 m	Different V_{up}	0.28 m/h > 0.85 m/h	0.28 m/h > 0.85 m/h
		0.28 m/h = 0.43 m/h	0.28 m/h = 0.43 m/h
		0.43 m/h = 0.85 m/h	0.43 m/h > 0.85 m/h
h_{sb} = 0.92 m	Different V_{up}	0.28 m/h = 0.43 m/h = 0.85 m/h	0.28 m/h = 0.43 m/h = 0.85 m/h
V_{up} = 0.28 m/h	Different h_{sb}	0.46 m = 0.92 m = 1.38 m/h	0.46 m = 0.92 m = 1.38 m/h

CONCLUSIONS

- TSS and VSS removal efficiencies around 90% have been achieved when $V_{up} \leq 0.43$ m/h (HRT ≥ 6 h) and $h_{sb} \geq 0.92$ m.
- TSS/VSS removal efficiencies were always higher than 70% when $h_{sb} \geq 0.92$ m, irrespective of the V_{up} applied.
- TSS/VSS removal efficiencies higher than 70% were always observed when $V_{up} \leq 0.43$ m/h. TSS/VSS removal efficiency was inversely proportional to V_{up}.
- SMA in the sludge bed was not affected by sludge discharges.
- Excess sludge discharge operations can be performed once every two years.

- A safe and efficient operation could be achieved in 4-m tall UASB reactors treating settled sewage under local conditions with h_{sb} between 1 (minimum h_{sb}) and 3 m (maximum h_{sb}) and V_{up} around 0.5 m/h.
- TSS/VSS removal efficiency could be a useful criterion to decide the right moment for sludge discharges.
- Results from this study only apply to settled (and quite diluted) sewage.

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CHAPTER 5

Two-step upflow anaerobic sludge bed (UASB) system for sewage treatment under subtropical conditions with posttreatment in waste stabilization ponds

ABSTRACT

A pilot-scale sewage treatment system consisting of two upflow anaerobic sludge bed (UASB) reactors in series followed by a system of five waste stabilization ponds (WSPs) was studied under subtropical conditions (mean sewage temperature = 23.0ºC). The use of two-stage anaerobic systems has been proposed to improve the retention and degradation of SS at low temperatures. The startup of the first UASB reactor was achieved in one month, without inoculum, at a hydraulic retention time (HRT) of 8.2 h, and an upflow velocity (V_{up}) of 0.48 m/h. A total chemical oxygen demand (COD_{tot}) removal efficiency of 89.2% was obtained in the two-step anaerobic system at HRTs of 6.4 + 5.6 h (V_{up} = 0.62 + 0.70 m/h), with 83.0 and 36.1% removal in the first and second steps, respectively. Fecal coliform removal in the whole system was 99.9999% (99.94% in the anaerobic steps and 99.98% in the WSPs). The system consistently complied with discharge standards for COD and fecal coliform. The performance of the system was not affected during the coldest period of the year, which lasted more than three months. COD balances over the two UASB reactors were calculated and a minimum set of data needed to build comprehensive COD balances over UASB reactors is proposed. Specific methanogenic activity (SMA), measured at 30°C, in the first and second UASB reactors was 0.12 and 0.04 gCOD-CH_4/gVSS.d (grams of methane-COD per gram of volatile suspended solids per day), respectively. The sludge growth rate in the first UASB reactor was 4.9% of the reactor volume per month. The excess sludge could be disposed of through only one or two discharges per year. Discharges should be performed in summer, when the sludge was found to be fully stabilized. Under the conditions of this work, the second anaerobic step was not necessary to achieve high COD removal efficiency, a good-quality effluent, and satisfactory sludge stabilization. One UASB reactor followed by WSPs could be a very efficient system for sewage treatment in subtropical regions.

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INTRODUCTION

In tropical countries, upflow anaerobic sludge bed (UASB) reactors treating sewage showed chemical oxygen demand (COD) removal efficiencies around 65%, with some reports of up to 80% in low loaded reactors (Wiegant, 2001). The hydraulic retention time (HRT) applied fluctuates around 6 h, aiming at an upflow velocity (V_{up}) of about 0.75 m/h in standard 4-m tall reactors (Wiegant, 2001). At lower temperatures, reported results differ widely, depending on factors such as sewage temperature and composition, operational parameters, type and dimensions of the reactor, and the amount and quality of the inoculum (see chapter 1). Removal efficiency decreases at lower temperatures (van Haandel and Lettinga, 1994). Analysis of data from several works reviewed in chapter 1 indicates that average COD removal efficiencies of 41.7, 52.8, and 69.1% have been observed at temperatures below 15ºC, between 15 and 22ºC, and above 22ºC, respectively. The relationship among temperature, COD removal efficiency, and HRT was studied by Zakkour *et al.* (2001). Two-stage anaerobic systems have been proposed as one of the ways to retain and degrade suspended solids (SS) from raw sewage at low temperatures (van Haandel and Lettinga, 1994; Wang, 1994; Elmitwalli, 2000).

Anaerobic sewage treatment systems, while achieving an acceptable removal efficiency, generally fail to comply with the discharge standard of 125 mgCOD/L established by Council Directive 91/271/EEC on Urban Waste Water Treatment, dictated by the European Union Council of Ministers (1991). Therefore, a posttreatment step is mandatory in most cases, not only to remove remnant COD, but also to remove nitrogen and phosphorus (when reuse is not possible), and fecal coliforms, the most commonly used indicator of pathogenic microorganisms. Waste stabilization ponds (WSPs), sometimes referred to as “polishing” ponds, are being studied as a posttreatment method for anaerobically treated sewage because they are among the most efficient and cost-effective methods (van Haandel and Lettinga, 1994; Cavalcanti, 2003). Sludge washed out from a UASB reactor was effectively trapped in a polishing pond, yet the sludge growth in the bottom of the pond was so low (including dead, settled algae), that desludging of the pond was deemed not necessary during its useful life-span (Cavalcanti, 2003). Therefore, an additional advantage of WSPs is that sludge discharges from UASB reactors (a major factor in operational costs) can be reduced or even eliminated. Local kinetic constants are necessary to accurately design WSPs. However, most of the WSPs in northern Argentina have been designed using extrapolations, adaptations, or regional constants (Liberal *et al.*, 1998).

COD balances over UASB reactors might be a useful tool to get insight into the flow of organic matter through the reactor, assess the performance of the process, validate methods and assumptions, and predict outputs. A COD balance is based on the fact that when a (relatively) constant average flow and load are applied to the reactor for a relatively long period of time, usually three times its solids retention time (SRT), and organic matter does not accumulate in the system, a “steady-state” is reached. At this point, the daily mass of influent COD is equal to the sum of the daily mass of COD leaving the system in one of several possible forms (methane, excess sludge, effluent COD, among others). Reports of COD balances in UASB reactors treating sewage have been scarce (Kalogo, 2000). However, some researchers have provided information about their systems that could lead to the formulation of COD balances (Wang, 1994; Elmitwalli, 2000). No references have been found for coupled COD balances in two-step UASB systems. van Haandel and Lettinga (1994) proposed a basis for the construction of COD balances over UASB reactors but a standardized methodology is still lacking.

Anaerobic sludges are often characterized by their specific methanogenic activity (SMA). SMA determination is useful to select seed sludges, determine organic loading rates, follow the development of the sludge, detect inhibitions and toxic effects, prevent accumulation of inert material in the sludge bed, and determine the sludge profile within anaerobic reactors (Soto *et al.*, 1993). SMA is measured in batch tests where a certain amount of sludge digests an easily biodegradable substrate under optimum environmental conditions. SMA is calculated from the maximum methane production rate.

In general, biological sludges need to be "stabilized" before they are suitable for reuse or final disposal. Sludge stabilization aims to reduce pathogens, eliminate offensive odors, and inhibit, reduce, or eliminate the potential for putrefaction (Metcalf and Eddy, 1991). Anaerobic digestion is the dominant sludge stabilization process. During anaerobic digestion, a substantial reduction in the amount of biodegradable organic matter can be achieved. The maximum amount of methane that can be produced by the sludge under anaerobic conditions represents the sludge anaerobic biodegradability, which is an excellent indicator of the sludge stability. The higher the anaerobic biodegradability the less stabilized the sludge. Ultimate stability should not change significantly with temperature, yet the degradation rate can vary considerably at different temperatures (Mahmoud, 2002). The U.S. Environmental Protection Agency defined three main aspects to be assessed before a sewage sludge is considered stable and safe enough to be applied to the land, namely the levels of pollutants (metals), the presence or absence of pathogens (i.e. disease causing organisms), and the degree of attractiveness of sewage sludge to vectors (vectors are animals and insects that might be attracted to sewage sludge and therefore, could transmit pathogenic organisms to humans) (EPA, 1993). Specific methods were recommended to meet the requirements set for each of the three criteria. The attractiveness of sludge to vectors is measured through the reduction in the content of volatile solids (VS) during the stabilization process. In a bench-scale anaerobic batch test, methane production and reduction of the VS are both a direct consequence of anaerobic degradation. Therefore, the determination of the VS reduction (attractiveness to vectors) and the measurement of the total amount of methane recovered (anaerobic biodegradability) should provide similar conclusions about the stability of the tested sludge.

The objectives of the present study were to assess the startup and subsequent operation of a two-step UASB system for sewage treatment under subtropical conditions, followed by five WSPs in series for posttreatment, and to build and contribute to the standardization of COD balances over UASB reactors.

MATERIALS AND METHODS

Pilot plant

Experiments were performed in the city of Salta, Argentina. The climate in the region is defined as subtropical with a dry season. Ambient temperature in the city and surroundings is 16.5 ± 0.2ºC (Arias and Bianchi, 1996). See chapter 2 for more details about the location.

The pilot plant (**Figure 1**) was installed at the city's main sewage treatment plant described in chapter 2. The dimensions of the first UASB reactor (R1) were the following: height = 3.95 m; diameter = 1 m; volume = 3.10 m^3. The second UASB reactor (R2) had the following dimensions: height = 3.95 m; diameter = 0.5 m; volume = 0.766 m^3. The dimensions of the WSPs were as follows: length = 3 m; width = 0.5 m; mean depth = 0.94 m; mean volume = 1.39 m^3; total volume = 6.97 m^3. Thirteen sampling ports (diameter = 3/4 in.) along both UASB reactors (R1 and R2) allowed liquid and sludge sampling. Up to 2 m high, ports were separated 0.20 m (10 ports); from 2 to 3.50 m, the distance between ports was 0.50 m (three additional ports). The last port was 0.45 m below the effluent exit.

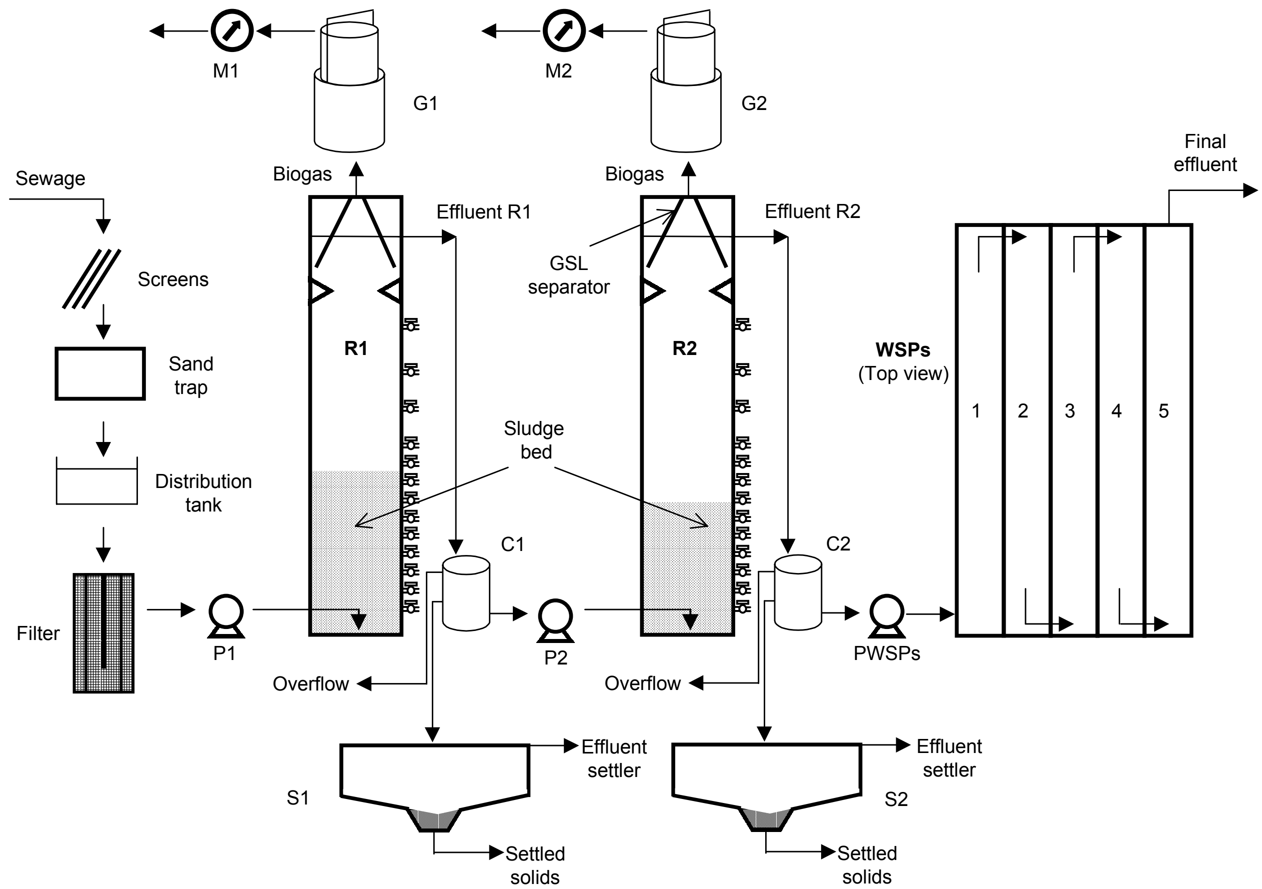

Figure 1. Schematic diagram of pilot plant (not to scale). R1: first UASB reactor; R2: second UASB reactor; WSPs: waste stabilization ponds (1-5); P1, P2, and PWSPs: peristaltic pumps; G1 and G2: gas accumulators (volume = 0.3 m^3); M1 and M2: gas meters; C1 and C2: intermediate pumping containers (volume = 10 L); S2 and S2: settlers (volume = 50 L); GSL = gas-solid-liquid.

Polyvinyl chloride (PVC) tubes and hoses (id = 1 in.) were used in the reactors for influent and effluent distribution. A transparent PVC tube (id = 3 mm) was used for the influent of the WSPs. Retention valves were installed in the reactors at the influent entrance to prevent sludge from flowing out during pump maintenance operations and power cuts. The influent of R1 was screened through a double 5-mm filter to prevent clogging, as suggested in Haskoning and WAU (1994). Influent was distributed in the reactors through one inverted inlet pipe located 5 cm from the bottom. Reactors, ponds, and gas accumulators were constructed of polyester reinforced with glass fiber at a local company (JJS Industrias Plásticas y Mecánicas). Watson Marlow 701 I/R, 621 I/R, and 313 S peristaltic pumps equipped with Marprene® tubing were used to feed R1, R2, and WSPs, respectively. Flow rates could be freely changed between a wide range in all units. Liquid was pumped in R2 and WSPs from two intermediate 10-L containers (C1 and C2) in which the effluents from R1 and R2 were discharged. Surplus was eliminated through overflow pipes. The containers were not tightly sealed, but they remained always closed to minimize stripping of dissolved methane. Biogas was collected in 0.3-m^3 gas accumulators and automatically measured in domestic gas meters (ABB ELSTER and Schlumberger Gallus 2000) operated by electric valves (Jefferson) and switches (Neumann CB 130).

The system was designed to operate at HRTs of approximately 6h, 4h, and 15 d in R1, R2, and WSPs, respectively. R1 was intended to be the main (primary and secondary) treatment step while R2 was included as a polishing step. Bypassing R2 for some time, the system was also operated as a single-stage UASB + WSPs. The HRT in R1 during startup was set to 8 h. When the startup was finished, HRT in R1 was decreased to the design value. A second startup at the design HRT of 6 h was also performed after one year. Both startup operations were performed in summertime, to minimize the possibility of reactor failure. The system (especially R1) was intended to be operating in (pseudo) steady state at the design HRTs by wintertime. Different hydraulic parameters were also applied to R2 during the operation. A detailed description of the operation during the whole experiment is presented in **Table 1** (left part). Partially digested sewage sludge from conventional anaerobic digesters, and anaerobic granular sludge from a UASB reactor treating settled sewage were used as inoculum for R1 and R2, respectively (15% [v/v]).

Raw and concentrated sewage

Raw sewage was submitted to preliminary treatment (screens and sand trap) and enriched with secondary sludge before being fed into the system. This "concentrated" sewage was roughly composed of 40% secondary sludge and 60% raw sewage (see chapter 2). SS in concentrated sewage were significantly higher than in raw sewage. In principle, the higher the SS concentration in sewage, the more difficult its treatment by anaerobic methods becomes, especially at low temperatures. Therefore, the increased concentration of SS in concentrated sewage was initially considered as a hindrance for the appropriate development of anaerobic digestion in the first anaerobic reactor. Secondary sludge recirculation was initiated as a routine operating procedure at the sewage treatment plant by the time the anaerobic pilot plant was due to startup. This operation had not been taken into account during the planning stage of the research. However, it provided with a type of sewage that represented an additional challenge. Not only was raw sewage treatment going to be attempted at (relatively) low temperatures, but this sewage also contained an unusually high proportion of SS. It was assumed that the anaerobic treatment of raw sewage would be at least as feasible as the treatment of concentrated sewage, considered as a more unfavorable scenario. Sewage temperature was measured before and after each unit (two UASB reactors and five WSPs) twice a week in grab samples with a digital thermometer (Keithley). Temperature in settled sewage (after 2 h in primary settlers) was also continuously recorded for more than two years with a thermograph Novasen 3752-5-S-C (temperature range: 0-50ºC). As reported in chapter 2, temperature was very similar in raw and settled sewage. Therefore, continuous temperature measurements in settled sewage were used as an indication of the temperature in raw sewage.

Sample analyses

Two times a week, composite samples were taken before and after each unit (0.5 L every 3 h for 24 h). Samples were kept at 4ºC until analyzed. Total COD (COD_{tot}), paper-filtered COD (COD_{filt}) (Schleicher & Schuell 595½ 4.4-µm paper filters), and membrane-filtered (dissolved) COD (COD_{dis}) (Schleicher & Schuell ME 25 0.45-µm membrane) were determined in the samples. Suspended and colloidal COD (COD_{sus} and COD_{col}) were calculated as (COD_{tot} - COD_{filt}) and

(COD_{filt} – COD_{dis}), respectively. The effluent of the WSPs was filtered through Whatman GF/C 1.2-µm glass microfiber filters to retain algae, and COD of the filtered sample was measured ($COD_{f1.2}$). Sludge COD was measured in crushed and diluted samples. Total SS (TSS) and volatile SS (VSS) in the sludge were determined in centrifuged samples, and TSS/VSS in the supernatant that could be retained by Schleicher & Schuell 589 4.4-µm ashless paper filters were included. Analyses were performed according to APHA *et al.* (1995) or using HACH® micromethods. Statistical comparisons and confidence intervals (CIs) were built at a level of significance (α) of 0.05 (5%).

Methanogenic activity tests

SMA of the inocula and the sludge was determined at 30°C as described in chapter 3. Parallel tests were conducted at 20ºC in a refrigerator equipped with an Incutrol® incubator. Sludge used in these tests was sampled from both UASB reactors (R1 and R2), a UASB reactor treating settled sewage described in chapter 3 (R3), and conventional anaerobic digesters fed with a mixture of primary and secondary (aerobic) sludge. Knowledge on the methanogenic activity at field temperatures is useful to assess the effect of temperature on SMA, to obtain the activity under conditions similar to those expected in the field, and for design purposes (i.e. calculate the biogas production in a full-scale facility). The SMA protocol described in DET (1994) requires the use of an excess substrate concentration (1.5 – 2.0 gCOD-acetate/L) to guarantee constant availability for anaerobic bacteria. It's reasonable to believe that for each type of sludge there is an optimum substrate concentration below which maximum methanogenic activity can not be reached, and above which inhibition and toxicity may reduce substrate consumption. The optimum concentration will depend, among other factors, on the type and strength of the wastewater on which the sludge was grown, and the actual substrate concentration in the reactor. Four tests have been conducted to find the substrate concentration at which maximum methanogenic activity was obtained, using different anaerobic sludges. Concentrations tested ranged from 0.20 to 1.50 gCOD-acetate/L (typically 0.50, 0.75, and 1.50 gCOD-acetate/L).

Stability tests

Bench-scale stability tests were performed in duplicate, sealed serum bottles in which a blend of sludge and water was digested at 30ºC in a temperature-controlled room for a certain period of time (at least two months). Water was added to avoid scum layer formation, reduce the frequency of methane measurements, and for safety reasons (bottles can explode if methane production is too high). No nutrients, substrate, nor buffer were added to the bottles. Bottles were not inoculated because methanogenic activity in all sludges analyzed was enough to degrade the samples. Tests were performed at 30ºC in a temperature-controlled room. Methane production was monitored in time through the displacement of a 5% NaOH solution. Stability tests were performed with sludges from both UASB reactors (R1 and R2). Parallel measurements were conducted with primary sludge and digested sludge to obtain reference points. Primary sludge came from settling tanks after 2 h of settling. Digested sludge was a mixture of primary and secondary sewage sludge digested for about 30 days at 30ºC in conventional anaerobic digesters. Anaerobic sludge from R3 was also analyzed. This reactor had been operating for more than 7 years and could be considered at steady state. Organic matter in the bottles at any time during the anaerobic degradation was described by the following equation:

$$COD_{(t)} = f_h COD_{(0)} \times e^{-k_h \cdot t} + (1 - f_h) COD_{(0)} \qquad (1)$$

where $COD_{(t)}$ = remaining organic matter at time t (gCOD/L); f_h = biodegradable fraction, or sludge anaerobic biodegradability (-); $COD_{(0)}$ = organic matter at the beginning of the test (time = 0) (gCOD/L); k_h = hydrolysis rate constant (d^{-1}); and t = time. All COD was assumed to be in particulate form. k_h was calculated as the slope of a straight line derived from equation 1 (Sanders, 2002):

$$\ln\frac{COD_{(t)} - (1 - f_h)COD_{(0)}}{f_h COD_{(0)}} = -k_h t \tag{2}$$

Term $COD_{(t)}$ was calculated as

$$COD_{(t)} = COD_{(0)} - CH_{4(t)} \tag{3}$$

where $CH_{4(t)}$ is the cumulative methane production (gCOD). The sludge anaerobic biodegradability was calculated with the following equation:

$$f_h = \frac{CH_{4(\infty)}}{COD_{(0)}} \tag{4}$$

where $CH_{4(\infty)}$ = total amount of methane produced at the end of the test (gCOD). This f_h can only be obtained after all COD is degraded, a process that can take several weeks, or even months. However, cumulative methane production at any time can be described by the following equation (Veeken and Hamelers, 1999; Mahmoud, 2002):

$$\mathrm{CH}_{4(t)} = f_h COD_{(0)}(1 - e^{-k_h \cdot t}) \tag{5}$$

Therefore,

$$f_h COD_{(0)} = \frac{CH_{4(t)}}{(1 - e^{-k_h \cdot t})} \tag{6}$$

Term $f_h COD_{(0)}$ is the amount of biodegradable organic matter present at the beginning of the test. An estimation of f_h can be obtained at any time t (before total degradation is attained) as

$$f_h = \frac{f_h COD_{(0)}}{COD_{(0)}} \tag{7}$$

Now, an estimated k_h is needed in equation 6. A value from previous tests with similar substrates and environmental conditions can be used. Iterating equations 2, 6, and 7, estimations of f_h and k_h can be obtained before the tests are finished. Insufficient methanogenic activity or the lack of indispensable nutrients for bacterial metabolism could be limiting factors in the anaerobic digestion process during stability tests. Consequently, hydrolysis parameters measured under these conditions may not reflect the intrinsic or specific values.

Organic matter in anaerobic sludges can be biomass (active anaerobic bacteria), degradable organics, or inert organics. Methane production during stability tests comes not only from hydrolysis and methanogenesis of the degradable part, but also from the decay of biomass and subsequent degradation of dead cells. In the final period of a stability test, when all biodegradable components have been consumed, methane can only be produced from the degradation of dead anaerobic bacteria. The first-order decay rate constant (k_{dec}) can then be calculated as described by Mgana (2003).

Knowing k_h and k_{dec}, and the total COD present at the beginning of the test, the degradation of organics and biomass (including newly formed biomass) can be calculated at any time with equation 1. In the case of biomass, k_h has to be replaced with the calculated k_{dec}. The anaerobic biodegradability of biomass and non-bacterial organic matter was assumed to be the same. It was also assumed that about 10% of hydrolyzed COD was converted to biomass (Mgana, 2003). All degraded COD should be recovered as methane, because hydrolysis and decay are limiting steps in the process of anaerobic digestion. Therefore, the original concentration of biomass and degradable components in the sludge could be estimated by fitting the cumulative gas production with results from equation 1 (Mgana, 2003). The fitting variable was the proportion of biomass in the sludge. Inert COD in the sludge was calculated as total COD minus biomass COD minus biodegradable COD.

Volatile solids reduction

Vector attraction reduction was estimated through the measurement of the reduction in VS during stability tests (EPA, 1993). Vector attraction is adequately reduced if the mass of VS in the sewage sludge is reduced by at least 38% during the treatment of the sludge. This percentage is the amount of VS reduction that is attained by anaerobic or aerobic digestion plus any additional VS reduction that occurs before the sludge leaves the treatment works, such as through processing in drying beds or lagoons, or by composting. Frequently, sludge has been recycled through the biological wastewater treatment section of a treatment works or has resided for long periods of time in the wastewater collection system. During this time, it undergoes substantial biological degradation. If the sludge is subsequently treated by anaerobic digestion for a period of time, it is adequately reduced in vector attraction. Because it will have entered the digester already partially stabilized, however, the VS reduction after treatment is frequently less than 38%. Under these circumstances, the vector attraction reduction can be demonstrated by testing a portion of the previously digested sludge in a bench-scale unit in the laboratory. Vector attraction reduction is demonstrated if after anaerobic digestion of the sludge for an additional 40 days at a temperature between 30 and 37°C, the VS in the sludge are reduced by less than 17% from the beginning to the end of the bench test.

Anaerobic sludges are partially or totally stabilized and VS reduction will most likely be lower than 17% in a 40-d long anaerobic batch test. VS reduction in primary sewage sludge will most likely be higher than 17% in 40 d. However, bench tests can be of help to determine the time needed for a 38% reduction in the VS content of primary sewage sludge (or any other unstable sludge) and therefore, the required digestion time in a full-scale facility.

COD balances

COD balances were based on the following equation:

$$I - E - G - D - B - W - L - S - C = 0 \qquad (8)$$

in which I is influent, E is effluent, G is gaseous methane, D is dissolved methane, B is sludge bed, W is sludge washout, L is scum layer, S is sulfate reduction, and C is a correction factor (all terms expressed in terms of mass of COD) (**Figure 2**).

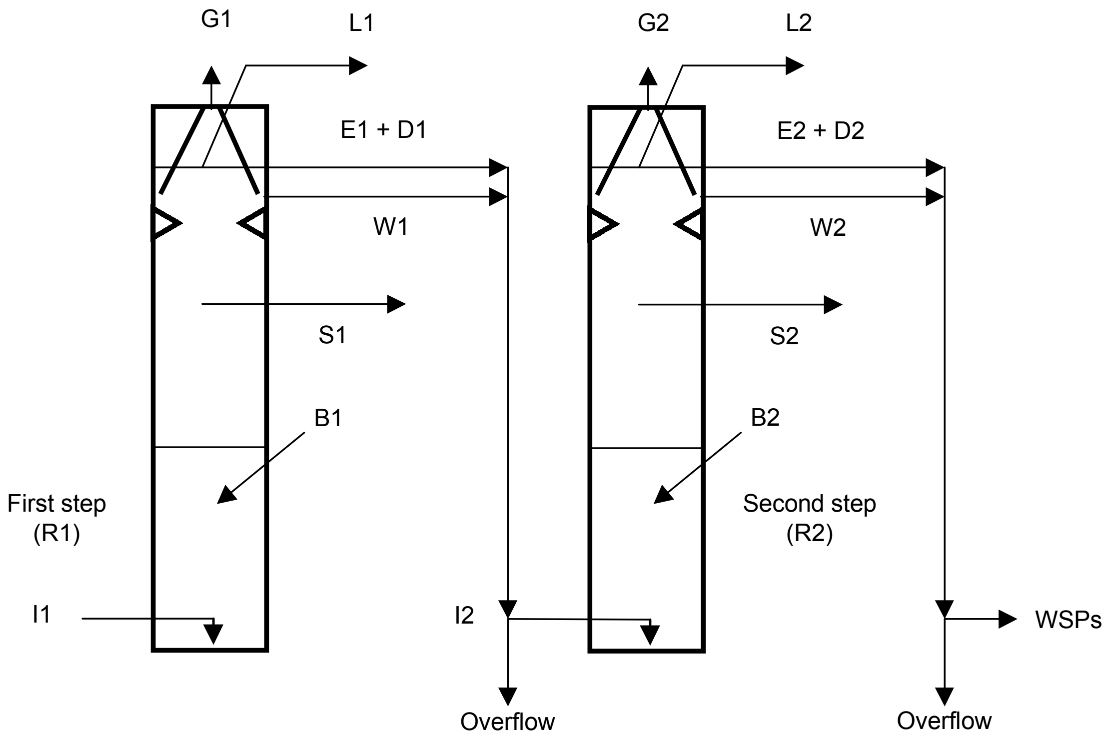

Figure 2. COD balances over two-step UASB reactors. I: influent; E: effluent; G: gaseous methane; D: dissolved methane (net value in R2); B: sludge bed; W: sludge washout; L: scum layer; S: sulfate reduction; WSPs: waste stabilization ponds.

Terms I and E were calculated by multiplying the COD concentration (in either influent or effluent) with the volume of sewage treated during the entire period. Term G was the methane produced during the period multiplied by a conversion factor to transform the volume under the conditions of the experiment to units of COD at STP conditions. The factor used was 2.23 kgCOD/m^3 of CH_4. Atmospheric pressure at the site was 0.866 atm. Methane content in the biogas was determined by stripping the CO_2 in a closed, up side down, serum bottle with a 5% NaOH solution, and collecting the displaced liquid in a graduated cylinder. The content of other gases in the biogas, like hydrogen sulfide, was neglected. Dissolved methane in the effluents (term D) was calculated according to Henry's law (Metcalf and Eddy, 1991), assuming that equilibrium concentrations were reached. Term B was the total amount of sludge accumulated in the reactor during the period. The increase in sludge bed was visually followed using the sampling ports. No excess sludge was discharged during the study. Sludge composite samples (an equal volume from all ports in which sludge could be extracted) were taken and analyzed for COD and solids. SRT in the UASB reactors was calculated according to van Haandel and Lettinga (1994) (see chapter 3). Term W was experimentally determined during period IV via two secondary settling tanks named S1 and S2 in

Figure 2 (V = 0.05 m^3; HRT = 2 h). Solids settled in the bottom were regularly withdrawn and quantified. The amount of sludge/solids lost from the reactors during the period was calculated based on the proportions of flow rate between the reactors and the settlers. Term L represented the scum layer formed on top of the reactors, which was regularly withdrawn, quantified, and analyzed with the same methodology used for the sludge. Term S represented the amount of sulfate reduction registered in the reactors. It was calculated as sulfate in the influent minus sulfate in the effluent, considering that 1 g of reduced sulfate is equivalent to 0.67 g of COD. Sludge withdrawn for analysis and days without operation (power cuts, maintenance, and so on) have also been accounted for in the balances.

Dimensioning of the ponds

Dimensioning of the ponds was based on the assumption that the death rate of pathogenic microorganisms follows first-order kinetics (Marais, 1974). Under this assumption, the HRT required in each pond of a certain number of completely-mixed ponds in series is given by the following equation:

$$HRT = \frac{\left({N_0}/{N_n} \right)^{1/n} - 1}{K} \tag{9}$$

where HRT = hydraulic retention time (d) in each of the ponds, assumed to be all equal; N_0 and N = concentration of pathogens in the influent to the first pond and the effluent of the last pond, respectively, expressed as the most probable number (MPN) of fecal coliform every 100 mL; n = number of ponds; and K = die-off constant at local temperature (d^{-1}). The value of K was calculated with the following relationship (derived from the equation of van't Hoff-Arrhenius):

$$K = K_{20} \times \theta^{(T-20)} \tag{10}$$

where K_{20} = die-off constant at 20ºC; θ = temperature coefficient (-); T = local sewage temperature (ºC). From studies performed in WSPs in the region, and data from literature, the following values were selected as inputs for equation 10: K_{20} = 1.5 d^{-1}; θ = 1.17; and T = 21ºC (average temperature during the coldest month of the year in a nearby full-scale anaerobic pond) (von Sperling, 1996; Liberal *et al.*, 1998). The World Health Organisation (WHO) recommends that, for unrestricted irrigation, treated sewage must have less than 1000 MPN/100 mL (WHO, 1989).

Assuming that raw sewage from the city of Salta contains about 1×10^7 MPN/100 mL, the removal efficiency required to reach the WHO recommendation would be 99.99%. Under the conditions described, it can be calculated that five completely mixed tanks in series could reach the removal efficiency required with an overall HRT of about 15 d (3 d in each pond).

The final dimensions of the ponds were based on: (a) the flow rate that could be provided by a peristaltic pump Watson Marlow 313 S, (b) a pond depth of 1 m, and (c) a relationship length/width = 6 in each pond (Liberal *et al.*, 1998).

RESULTS AND DISCUSSION

Sewage

The basic composition of the concentrated sewage used during the experiments was presented and discussed in chapter 2. The anaerobic biodegradability of concentrated sewage was 74.1 and 62.7% at 30 and 20ºC, respectively (anaerobic biodegradability tests were also described in chapter 2). Mean temperature in raw sewage was 23.0 ± 0.3ºC (based on continuous measurements performed in settled sewage over two years). Mean monthly sewage temperatures below 20ºC were observed for up to 4 months in a row, with a mean monthly minimum of 17.2 ± 0.7ºC, and a mean daily minimum of 12.6 ± 2.4ºC. More details about temperature measurements were provided in chapter 2.

Startup

Results were divided in periods. Periods I to IV correspond to the first year of operation; periods V and VI to the second year (**Table 1**). Temperatures reported in different points of the system are averages from grab samples. Period I was the startup, when HRTs of 8.2 h, 4.0 h, and 14.5 d were applied in R1, R2, and WSPs, respectively. The sludge volume index (SVI) of the inocula used in R1 and R2 were 16.7 ± 6.2 and 10.7 ± 5.0 mL/g, respectively. These values indicate good settling characteristics (Uemura and Harada, 2000; Halalsheh, 2002). However, the inoculum of R1 was completely washed out during the first week of operation, probably due to a sudden sludge flotation. The reactor was not inoculated again and therefore, the startup was eventually performed without inoculum. The inoculum of R2 was not washed out, even though the applied V_{up} was higher during startup, probably because settleability of the inoculum was better, and biogas production was lower than in R1. The startup period in R1, considered to be the most critical step of the system, lasted about 1 month. The startup was considered over when all the conditions proposed in Noyola Robles (1994) to indicate stable operation and/or to decide on an increase of organic load were met, namely:

(a) COD removal efficiency reached more than 80% of the design value. The design value considered was the removal efficiency required to reach discharge standards for COD_{tot} (125 mg/L). With an influent concentration of about 400-450 mgCOD_{tot}/L, the minimum removal efficiency needed would be around 70%. Therefore, a removal efficiency of at least 56% must be reached during startup. As COD_{tot} removal in R1 was already around 80% by the end of the first month of operation, this condition was easily met.

(b) The ratio between alkalinity measured at pH 5.75 and pH 4.3 was higher than 0.7 in the effluent. On average, total alkalinity (at pH 4.3) was slightly higher in the effluent than in the influent (**Figure 3**).Volatile fatty acids (VFA) remained always very low in the effluent since the very beginning of the operation (**Figure 3**). VFA were remarkably similar in influent and effluent, probably because their anaerobic biodegradability was limited. In any case, accumulation of VFA was not detected in the reactor at any time, indicating that methanogenesis was never exceeded.

(c) Biogas production was about 0.1 Nm^3/kg$COD_{removed}$ (letter N indicates that volume is expressed at STP conditions) with a very stable CH_4 content of 90%. Maximum possible methane production from organic matter is 0.350 Nm^3/kg$COD_{removed}$ (van Haandel and Lettinga, 1994). Assuming 70% methanogenesis of removed COD (based on results from biodegradability tests reported in chapter 2), the maximum expected methane production at steady state would be around 0.25 $Nm^3/kgCOD_{removed}$. Therefore, 40% of the expected methane recovery in steady state was reached during startup.

At the end of the first month, other criteria for "steady state" set in the literature were also met, namely that the operation time was more than 10 times the HRT (and more than 2 weeks) (Noyola *et al.*, 1988), and variations in effluent concentration were lower than ± 10% (Polprasert *et al.*, 1992). Elmitwalli (2000) and Mahmoud (2002) considered these criteria satisfactory. The steady state criteria were only based on performance results, and not on an analysis of the characteristics of the sludge. In fact, at the end of the startup period the sludge bed was not yet fully established. A real steady state would only be achieved in the sludge bed, and consequently in the reactor, if the operation period is at least three SRTs (van Haandel and Lettinga, 1994).

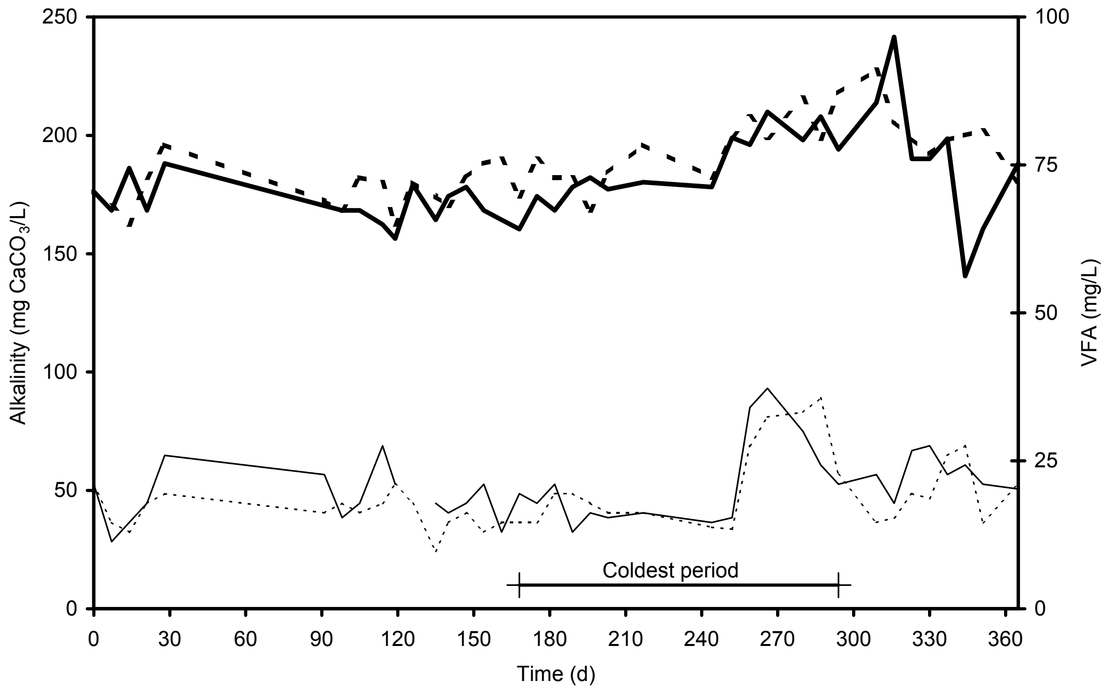

Figure 3. Alkalinity (thick lines) and VFA (thin lines) as a function of time in R1 for the first year of operation. Full line: influent; dotted line: effluent. The coldest period of this year (99 d) is indicated.

Operation

Table 1 (right) shows COD fractions in the influent to the entire system, hydraulic conditions applied, and COD removal efficiencies for each experimental period. Mean organic loading rates (OLR) applied during the first year were 1.48 and 0.43 $kgCOD_{tot}/m^3{}_{reactor}.d$ in R1 and R2, respectively. Mean V_{up}s applied during the first year were 0.56 ± 0.02 and 0.82 ± 0.02 m/h in R1 and R2, respectively. Calculated SRTs were 169 d in R1 and 404 d in R2. These values are more than enough to achieve sufficient hydrolysis and methanogenesis at working temperatures in both reactors, and were in agreement with the predictions made in chapter 2 using the equation proposed by Zeeman and Lettinga (1999). When the startup was considered over, the system was operated at the same hydraulic conditions for two more months (period II) to guarantee a stable performance. COD_{tot} removal in R1 stayed constant around 80% during period II. However, removal in R2 dropped to less than 10%, while some washout of seed sludge was observed (negative removal of COD_{sus}).

In period III, HRT was decreased to about 6 h in R1. COD_{tot} removal in R1 did not decrease in this period, but increased in R2, due to a higher COD_{tot} concentration in the influent. COD_{sus} removal efficiency decreased in R1 during period III due to the higher V_{up} applied, but increased consequently in R2. Negative removal of COD_{col} was observed in R2 during this period.

In period IV, HRT was increased in R2, but was maintained at previous values in R1. During period IV, R1 was considered at (pseudo) steady state, although only one SRT had elapsed since the first day of operation. High COD_{tot} removal efficiency was consistently recorded in R1. COD_{col} and COD_{dis} removal efficiency increased steadily in R1 since the beginning of the experiment, and stabilized in period IV. A clear trend was not observed in R2 for these fractions. COD_{sus} removal efficiency in R1 during period IV recovered to the high values observed in period II. Inversely, the removal of this fraction decreased in period IV in R2. The coldest period of the year (99 d) fell within period IV, when mean sewage temperature dropped to 19.4 ± 0.3 (n = 30). Hydraulic conditions during this colder period were HRT = 6.5 ± 0.4 h and V_{up} = 0.61 ± 0.03 m/h. Removal efficiencies attained in R1 were 81.3% for COD_{tot}, 92.2% for COD_{sus}, 66.2% for COD_{col}, and 14.5% for COD_{dis}. Removals were similar to those recorded in summer (period II), even though a lower HRT was applied in the cold period. Methane recovery during winter was similar to values observed in previous periods, and no increase in VFA concentration was detected in the effluent (**Figure 3**).

The sludge of R1 was completely discharged after one year, and this reactor was started up again at an HRT = 6.2 h (period V). No discharge of sludge was performed in R2. Second startup in R1 was also accomplished in about one month, although applied V_{up} was 30% higher than during the first startup. This indicates that UASB reactors can be operated in subtropical regions at a design HRT of about 6 h since the beginning, and there is no need to apply a higher HRT during startup. A slightly lower COD removal efficiency was observed in R1 in period VI, compared to homologous period IV from the first year, probably due to different concentrations in the influent to the system. However, a removal higher than 70% was observed during this entire period, which lasted almost one entire year. This value was in the range of results obtained with UASB reactors treating raw sewage in tropical regions. The effluent of R1 consistently complied with municipal, provincial, and european discharge standards for COD_{tot} (Municipalidad de Salta, 2000; SeMADeS, 2001; European Union Council of Ministers, 1991) (**Figure 4**). Algae grown in the ponds were responsible for the increase in COD_{sus} in the final effluent, but it still complied with discharge standards, making a final clarification step unnecessary. COD_{col} and COD_{dis} remained rather constant after the first anaerobic step. Reasonable removal efficiencies and effluent concentrations were observed in R1 during both startup periods (performed without inoculum), suggesting that reactor performance is unlikely to be affected by heavy sludge discharges. The presence of R2 contributed to reduce even further the effluent concentration, but it was apparent that this reactor was redundant even when R1 was under stressing conditions (e.g. startup periods, wintertime). For this reason, R2 was no longer operated in period VI. Removal efficiency in the WSPs increased in period VI due to the higher influent concentration. However, the final effluent concentration was similar to that observed in period IV, confirming that a second UASB step was not required.

Table 1. Temperature, hydraulic conditions, influent concentration, and removal efficiencies in system during different periods[a].

Period	Days	Step	Temperature (°C)[b]	Hydraulic conditions HRT	Hydraulic conditions V_{up} (m/h)	COD fraction	Influent (mg/L)	Removal efficiency (%)[c] Total	R1	R2	R1+R2	WSPs
I	1-33 (33 d)	R1	23.3 ± 0.2	8.2 ± 0.1 h	0.48 ± 0.01	COD_{tot}	367.1 ± 42.2	78.5	62.2	27.2	72.5	21.7
		R2	23.3 ± 0.5	4.0 ± 0.1 h	0.99 ± 0.01	COD_{sus}	270.4 ± 66.7	88.1	85.0	5.9	85.9	15.2
		WSPs	24.2 ± 1.6	14.5 ± 1.4 d		COD_{col}	18.3 ± 8.7	42.1	-1.6	18.8	17.5	29.8
						COD_{dis}	63.1 ± 12.2	50.6	13.0	29.0	38.2	20.0
II	33-96 (63 d)	R1	24.0 ± 0.1	8.1 ± 0.1 h	0.49 ± 0.01	COD_{tot}	354.1 ± 61.4	77.7 (86.1)	80.4	8.1	82.0	-23.7 (22.6)
		R2	23.3 ± 0.5	4.2 ± 0.1 h	0.93 ± 0.02	COD_{sus}	288.8 ± 72.6	85.4	94.5	-18.2	93.5	-126.1
		WSPs	25.6 ± 1.1	15.0 ± 1.2 d		COD_{col}	22.4 ± 15.1	37.7	16.8	32.6	43.9	-10.9
						COD_{dis}	43.4 ± 6.8	35.1	26.9	3.5	29.5	8.0
III	96-166 (70 d)	R1	23.2 ± 0.5	6.1 ± 0.1 h	0.65 ± 0.01	COD_{tot}	420.8 ± 96.5	78.0 (89.0)	80.0	39.7	87.9	-82.5 (8.6)
		R2	23.3 ± 0.7	4.1 ± 0.1 h	0.96 ± 0.02	COD_{sus}	269.1 ± 52.3	84.9	84.0	59.2	93.5	-131.0
		WSPs	23.7 ± 1.9	15.7 ± 2.2 d		COD_{col}	43.1 ± 26.0	61.9	89.8	-106.0	79.0	-80.8
						COD_{dis}	56.8 ± 19.3	33.6	32.9	25.7	50.1	-33.1
IV	166-324 (158 d)	R1	20.3 ± 0.4	6.4 ± 0.2 h	0.62 ± 0.02	COD_{tot}	472.5 ± 50.6	79.8 (91.3)	83.0	36.1	89.2	-86.3 (19.7)
		R2	20.1 ± 0.6	5.6 ± 0.1 h	0.70 ± 0.01	COD_{sus}	360.1 ± 74.7	85.7	92.3	50.4	96.2	-274.5
		WSPs	18.8 ± 1.4	15.6 ± 0.8 d		COD_{col}	37.9 ± 13.8	70.3	72.6	14.7	76.6	-26.9
						COD_{dis}	45.2 ± 8.8	38.2	22.8	29.4	45.5	-13.5
V	324-356 (32 d)	R1	23.2 ± 0.2	6.2 ± 0.2 h	0.63 ± 0.02	COD_{tot}	328.3 ± 40.5	66.2 (83.7)	66.4	36.7	78.8	-59.3 (23.1)
		R2	22.9 ± 0.8	5.5 ± 0.1 h	0.71 ± 0.01	COD_{sus}	243.3 ± 61.9	75.8	84.8	26.3	88.8	-116.5
		WSPs	23.9 ± 2.2	19.1 ± 5.0 d		COD_{col}	39.7 ± 10.7	50.1	44.8	32.4	62.7	-33.8
						COD_{dis}	54.4 ± 14.8	37.7	17.3	23.1	36.4	2.0
VI	356-699 (343 d)	R1	24.1 ± 0.4	6.1 ± 0.1 h	0.64 ± 0.01	COD_{tot}	362.9 ± 43.6	71.8 (86.9)	73.9			-8.4 (49.5)
		WSPs	25.7 ± 2.2	17.1 ± 2.8 d		COD_{sus}	282.1 ± 49.7	81.5	85.4			-26.3
						COD_{col}	46.0 ± 9.1	52.5	53.3			-1.8
						COD_{dis}	59.8 ± 6.2	47.3	43.4			6.9

[a]Mean values ± CIs are reported. [b]Influent temperature for R1 and R2; average temperature for the five WSPs. [c]Calculated from average concentrations in each period. Values in parenthesis were based on algae-free final effluent.

Chernicharo *et al.* (2001) and Cavalcanti (2003) shown that WSPs, designed to remove fecal coliform, are not fully exploited with respect to COD removal. Posttreatment in WSPs is necessary anyway to disinfect the effluent and reach discharge standards for pathogenic microorganisms. On the other hand, for a pond to work properly, a certain amount of biodegradable organic matter is required in the influent. Therefore, in a full-scale system, the UASB reactor could be operated at an HRT lower than 6 h, and the extra COD in the effluent could be treated in a series of WSPs designed for pathogen removal.

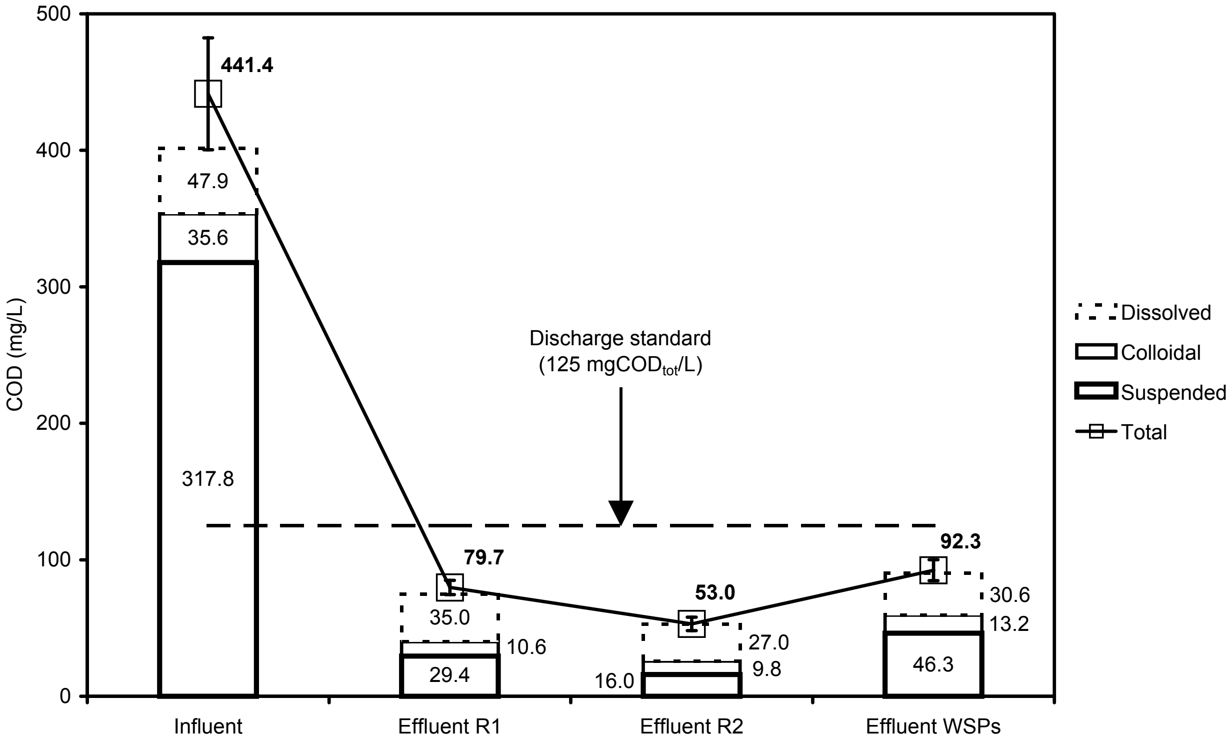

Figure 4. COD fractions through the system during the first year of operation (results from periods II, III, and IV; startup period I was not included). The sum of suspended, colloidal and dissolved COD is slightly different from total COD values because the latter come from a larger set of data. CIs are shown as error bars.

Methanogenic activity

SMA in R1 was significantly higher than in R2, possibly due to differences in influent concentration, and the lack of readily biodegradable COD in the influent of R2 (**Table 2**). SMA observed in R1 was in the range reported in literature (van Haandel and Lettinga, 1994; Haskoning and WAU, 1994). SMA of the inocula was 0.078 and 0.033 gCOD-CH_4/gVSS.d for R1 and R2, respectively. SMA did not change significantly in R2 during operation. Comparison of several tests conducted along a prolonged period of time with sludges from different sources showed that SMA was significantly higher at 30ºC than at 20ºC (paired comparisons of means; α = 0.01) (**Table 3**). Fluctuations in SMA at both temperatures are due to the fact that samples came from different reactors, and from different sampling ports in the reactors. On average, methanogenic activity at 20ºC was 65% of the activity at 30ºC, assumed to be the maximum.

In previous works, it was reported that sludge from UASB reactors treating settled sewage was stratified, containing layers with different SMA and VSS content; an inverse relationship between SMA and VSS concentration was found (this thesis, chapter 3). Stratification was not observed in R1 at any time, probably due to higher biogas production. Stratification was neither observed in R2.

Tests performed with substrate concentrations between 0.75 and 1.25 gCOD-acetate/L rendered maximum methanogenic activity in all cases (raw data not shown). In none of these tests maximum methanogenic activity was observed at 1.50 gCOD-acetate/L. These results suggest that the substrate concentration needed to obtain the maximum possible methanogenic activity can not be completely standardized, but rather it should be previously determined for each type of sludge. Halalsheh (2002) reported that a high acetate concentration of 1 g/L (compared to that in the reactor) could have caused toxicity in SMA tests.

Table 2. SMA and VSS in the sludge bed from UASB reactors fed with different types of sewage[a].

Reactor	Influent concentration (gCOD/L)[b]				SMA	VSS
	COD_{tot}	COD_{sus}	COD_{col}	COD_{dis}	(gCOD/gVSS.d)	(gVSS/L of sludge)
R1	432,7 ± 36,9	311,9 ± 39,4	33,4 ± 9,4	49,8 ± 6,4	0,121 ± 0,013	24,1 ± 5,0
R2	86,6 ± 6,8	30,8 ± 5,6	11,6 ± 4,3	37,6 ± 5,0	0,037 ± 0,043	30,5 ± 2,6

[a] Mean values ± 95% CIs are provided. [b] Results from the first year of measurements (n = 77).

Table 3. SMA obtained at 20 and 30ºC with different anaerobic sludges.

Test	Sampling date	Sludge sample		VSS (g/L)	SMA ($gCOD-CH_4/gVSS.d$)	
		Reactor	Influent to the reactor		30ºC	20ºC
1	02-mar-01	R3	Settled sewage	39,1	0,030	0,026
2	03-may-01	R3	Settled sewage	18,2	0,117	0,053
3	29-jun-01	R3	Settled sewage	26,6	0,069	0,030
4	28-aug-01	R3	Settled sewage	27,6	0,095	0,066
5	16-oct-01	AD[a]	Sewage sludge	29,3	0,036	0,035
6	14-nov-01	AD	Sewage sludge	19,4	0,166	0,147
7	14-nov-01	R3	Settled sewage	63,3	0,043	0,035
8	05-jan-02	R3	Settled sewage	54,6	0,141	0,119
9	16-jan-02	R1	Concentrated sewage	24,14	0,109	0,041
10	16-jan-02	R2	Anaerobically-treated sewage	31,9	0,015	0,008
11	12-feb-02	R3	Settled sewage	56,3	0,087	0,066
12	18-feb-02	R1	Concentrated sewage	28,5	0,130	0,071
13	18-feb-02	R2	Anaerobically-treated sewage	29,2	0,059	0,065
14	18-feb-02	R3	Settled sewage	13,1	0,158	0,069
15	15-apr-02	R3	Settled sewage	58,4	0,059	0,034
16	18-oct-02	R3	Settled sewage	58,8	0,048	0,016
			Mean ± CI:	36,2 ± 8.1	0,085 ± 0.023	0,055 ± 0.018

[a]AD = conventional anaerobic sludge digester.

Stability

Methane production started immediately after the bottles were closed, indicating that there was enough methanogenic activity to promote the degradation of the samples in all types of sludges tested. Results presented in **Table 4** show that anaerobic biodegradability was lowest (and therefore stability was highest) in sludge from R2, probably due to the extremely low organic loading rate applied to this reactor. The second most stable sludge was summer sludge from R1. Sludge from R3 ranked third when anaerobic biodegradability was expressed in gCOD-CH_4/gCOD of sludge. However, digested sludge ranked third when anaerobic biodegradability was expressed in gCOD-CH_4/L of sludge. These two sludges presented similar anaerobic biodegradability expressed in gCOD-CH_4/gVSS of sludge. Winter sludge from R1 was the least stable of all treated sludges tested. As expected, maximum anaerobic biodegradability (lowest stability) was observed in primary sludge.

The COD/VSS ratio observed in sludge from different reactors was higher than the average ratio of 1.48 gCOD/gVSS normally reported for biological sludge (van Haandel and Lettinga, 1994) (**Table 4**). However, sludges analyzed in this work included not only microorganisms, but also undegraded organic matter, inert organic matter, and inert solids. The presence of undegraded lipids in the sludge could explain high COD/VSS values, as long as lipids are known to have a ratio of 2.91 gCOD/gVSS (Sayed, 1987). Lipids are degraded during the stabilization of the sludge, explaining lower ratios observed for more stable sludges. Chaggu (2004) also observed high COD/VS ratios in sludge of different ages from pit-latrines. Results expressed in terms of VS, instead of VSS, are not expected to differ significantly because the contribution of dissolved COD potentially present in the sludge (in principle, equal to the COD concentration in the liquid phase, which is always lower than 0.1 g/L) is negligible compared to the total COD concentration in the sludge (up to 60 g/L).

Stability observed in summer sludge from R1 was similar to that reported by Mahmoud (2002) in primary sewage sludge after 15 days of stabilization in a stirred reactor at 35ºC (for this comparison, results were converted to gCOD-CH_4/g of VS). It is interesting to notice that summer sludge from R1 was more stable than digested sludge from conventional sludge digesters. Bogte *et al.* (1993) reported that, under moderate temperature conditions, material settled in a UASB-septic tank during winter was degraded in summer. Under the conditions of the experiments in the present study, further stabilization of the (summer) sludge from R1 in a parallel, heated anaerobic digester, as proposed by Mahmoud (2002) for Middle East conditions (mean winter sewage temperature of about 15ºC) is not necessary. The sludge can be sufficiently stabilized within the UASB reactor. Very high sludge stability was reported in UASB reactors treating sewage in Cali (Colombia), and Kanpur (India) (van Haandel and Lettinga, 1994), with anaerobic biodegradability values of 0.19 and 0.20 gCOD-CH_4/gVSS, respectively.

Measured k_h values in primary and digested sludge were similar to those reported by Mahmoud (2002) in primary sludge digested from 10 to 30 d at 35ºC (0.11 d^{-1}). Low k_h values were observed in all anaerobic sludges. Anaerobic biodegradability values estimated with the iteration described in Materials and Methods were close to the values eventually measured at the end of the tests. Estimations were obtained in less than two weeks in most tests, while the average length of the complete tests was 103 ± 21 d. Observed k_{dec} were in the range of those reported in literature (Batstone *et al.*, 2002; Mgana, 2003). The fraction of biomass calculated as described in Materials and Methods that best fitted methane production data was similar for all stabilized anaerobic sludges (R1 in summer, R2, and R3) (**Figure 5**).

Table 4. Results from stability tests conducted with different sludges[a].

Sludge source	Test length (d)	Sludge composition			Anaerobic biodegradability			k_h (d^{-1})	k_{dec} (d^{-1})
		gVSS/L	gCOD/L	gCOD/gVSS	gCOD-CH_4/L	gCOD-CH_4/gVSS	gCOD-CH_4/gCOD		
R1 (summer)	105 ± 40	22.9 ± 3.8	50.7 ± 8.5	2.15	8.7 ± 1.6	0.38 ± 0.06	0.17 ± 0.03	-0.04 ± 0.01	-0.03 ± 0.01
R1 (winter)	75	26.2	56.3	2.22	18.1	0.69	0.32	-0.06	-0.04
R2	103 ± 39	31.7 ± 16.4	49.5 ± 19.4	1.56	4.7 ± 1.5	0.16 ± 0.03	0.11 ± 0.05	-0.03 ± 0.01	-0.03 ± 0.01
R3	129 ± 59	32.0 ± 19.6	59.7 ± 15.7	1.86	12.7 ± 6.4	0.42 ± 0.13	0.19 ± 0.08	-0.11 ± 0.05	-0.04 ± 0.01
Primary sludge	117 ± 92	23.5 ± 5.9	50.5 ± 7.4	2.15	25.6 ± 4.3	1.11 ± 0.20	0.51 ± 0.04	-0.10 ± 0.03	-0.04 ± 0.01
Digested sludge	68 ± 8	24.7 ± 7.2	39.5 ± 13.6	1.60	10.2 ± 2.1	0.42 ± 0.07	0.27 ± 0.05	-0.10 ± 0.03	-0.05 ± 0.04

[a]Means ± 95% CI are provided unless only one value available.

Table 5. Total and volatile solids (TS, VS) in different sludges before and after stability tests[a].

Sludge source	Before			After			Reduction (%)		
	TS (g/L)	VS (g/L)	VS/TS	TS (g/L)	VS (g/L)	VS/TS	TS	VS (total)	VS (in 40 d)
R1 (summer)	53.6 ± 9.4	23.2 ± 1.8	0.44 ± 0.11	49.5 ± 17.3	18.3 ± 5.9	0.37 ± 0.01	8.5	20.5	12.0
R1 (winter)	66.4	34.2	0.52	62.4	28.2	0.45	6.0	17.6	14.3
R2	38.7 ± 0.03	19.7 ± 1.0	0.51 ± 0.03	32.5 ± 6.9	15.3 ± 3.4	0.47 ± 0.01	16.0	22.3	14.7
R3	50.4	27.5	0.55	41.3	20.6	0.50	18.1	25.2	19.7
Primary sludge	55.4 ± 7.3	31.0 ± 3.7	0.56 ± 0.01	49.4 ± 1.5	19.2 ± 11.2	0.39 ± 0.22	10.5	39.1	37.3
Digested sludge	59.4 ± 19.4	29.9 ± 9.0	0.51 ± 0.05	49.9 ± 17.5	21.5 ± 7.1	0.43 ± 0.06	16.3	28.5	26.5

[a]Mean ± 95% CI is provided unless only one value available.

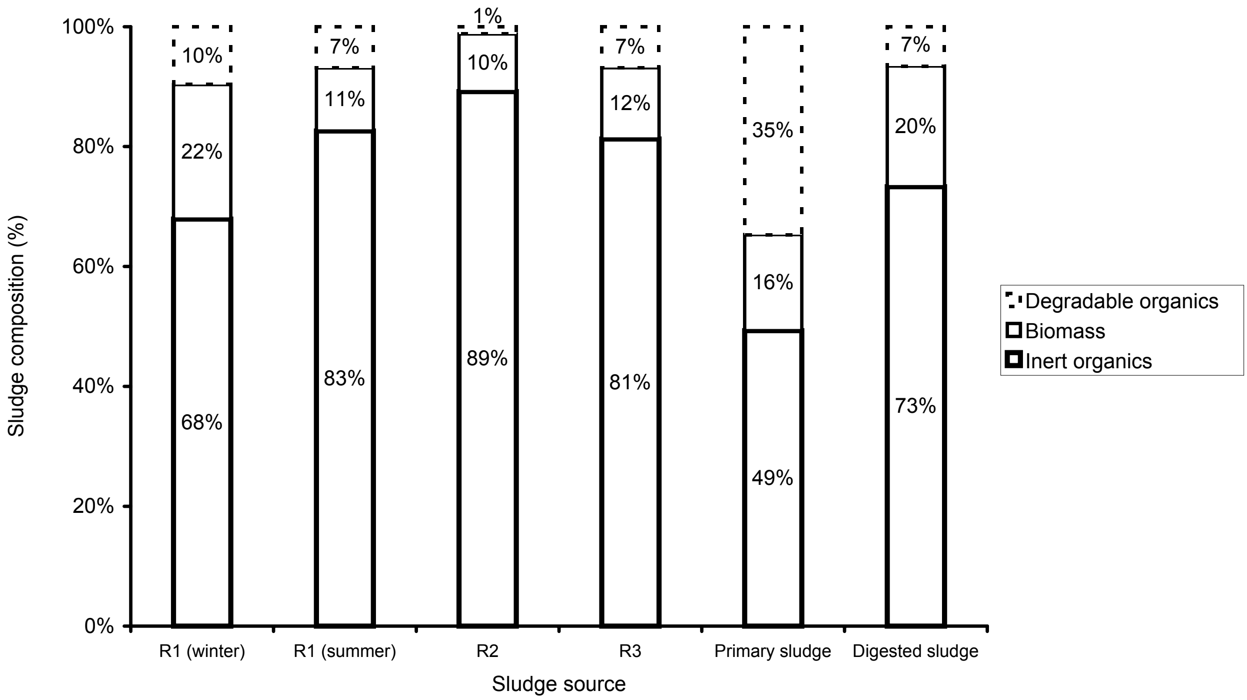

Figure 5. Sludge composition calculated by fitting experimental methane production data. The fitting variable was the proportion of biomass in the sludge.

A higher proportion of biomass fitted the results in winter sludge from R1, probably due to the accumulation of biomass from recirculated secondary sludge. The calculated composition of winter sludge from R1 was comparable to that of digested sludge. A high proportion of primary sludge was biodegradable organic matter, but nonetheless the best fit was obtained with a relatively high fraction of biomass. A clear explanation for this could not be found, although the presence of a significant amount of biomass could not be ruled out in primary sludge when the retention time in the settlers is prolonged for practical reasons. Indeed, a lot of biogas production was regularly observed in primary settlers. Proportions of inert organics, biomass, and degradable organics calculated for all types of sludges are in general agreement with stability results. After about one month of testing (27.3 ± 3.7 d), degradable organics were consumed in all sludges, and methane production could be explained only by biomass decay.

VS reduction observed in anaerobic sludges at the end of the tests was slightly higher than the 17% standard set by EPA (1992) for a stable sludge because the tests were longer than the 40-d digestion period required by the standard (**Table 5**). The VS reduction in 40 d, estimated from methane production data under the assumption that VS reduction is a direct consequence of biological degradation, was less than 17% in all anaerobic sludges (last column in **Table 5**). Therefore, these sludges could be considered stabilized under EPA standards, at least with respect to vector attraction reduction. VS reductions in primary and digested sludge were higher than 17% after 40 d of digestion, meaning that these sludges were not stable enough when the tests started. VS reduction observed in primary sludge was lower than that commonly reported for anaerobic digesters used for the stabilization of sewage sludge (45-60%) (Metcalf and Eddy, 1991). The amount of VS was always higher than the amount of VSS in all samples analyzed for both parameters. Discrepancies observed between **Table 4** and **Table 5** for sludges from R2 and R3 are due to the fact that only a fraction of the total number of samples was also analyzed for VS.

The methods used to measure stability, namely the anaerobic biodegradability method (stability tests) and the VS reduction method, both detected differences between the sludges tested. With both methods, sludge from R2 and summer sludge from R1 were among the most stabilized sludges, while the primary sludge was the least stabilized; stability in sludge from R3 and in the digested sludge was in between (**Table 6**). Some discrepancies were observed between the methods, especially concerning winter sludge from R1. However, only one test was performed with winter sludge, and these results have to be considered only preliminary. Stability tests allow the determination of hydrolytic parameters and give a clear idea of the course of the digestion process. On the other hand, VS reduction can be determined by two simple determinations before and after the 40-d period required by the EPA standard. The use of one method, the other, or both will depend on the objectives of the study, and practical considerations like availability of proper equipment and personnel to carry out the measurements. A standard procedure for stability tests is still lacking and comparison of results reported in literature can be equivocal (Mgana, 2003). Misleading conclusions could be drawn if anaerobic biodegradability is expressed in different units. A sludge stability standard, preferably expressed in gCOD-CH_4/gVSS, or gCOD-CH_4/gCOD, should be established. All in all, the EPA standard may be enough to determine sludge stability in most practical cases.

Table 6. Sludges ranked according to their relative stability for different methods.

Ranking (decreasing stability)	Method			
	Anaerobic biodegradability tests[a]			VS reduction[b]
	(gCOD-CH_4/L)	(gCOD-CH_4/gVSS)	(gCOD-CH_4/gCOD)	(%)
1	R2	R2	R2	R1 (summer)
2	R1 (summer)	R1 (summer)`	R1 (summer)	R1 (winter)
3	Digested sludge	Digested sludge	R3	R2
4	R3	R3	Digested sludge	R3
5	R1 (winter)	R1 (winter)	R1 (winter)	Digested sludge
6	Primary sludge	Primary sludge	Primary sludge	Primary sludge

[a] Results from **Table 4**.
[b] Results from **Table 5**.

Sludge management

During the first year of operation, sludge production in R1 was 0.08 kgCOD/kgCOD$_{removed}$ and represented about 4.9% of the reactor volume per month (v/v). Sludge production in this reactor was lower than that reported by Mahmoud (2002) for a combined system UASB- sludge digester (0.21 kgCOD/kgCOD$_{removed}$), probably because R1 operated at a higher temperature, and influent concentration was significantly lower. Sludge production in R2 was 1.8% of the reactor volume per month. However, sludge production per unit of removed COD in this reactor was 0.39 kgCOD/kgCOD$_{removed}$, higher than in R1. This could be attributed to accumulation of the increased proportion of inert organic matter present in the influent, as most of the anaerobically biodegradable organics were retained (and degraded) in the first step. Sludge COD was 56.2 ± 8.9 (n = 5) and 45.0 ± 3.0 (n = 2) g/L of sludge in R1 and R2, respectively. Anaerobic sludge from UASB reactors treating raw sewage can rise close to the effluent exit without compromising the quality of the effluent and the treatment efficiency (Cavalcanti, 2003). Therefore, frequent sludge discharges may not be necessary, reducing the need of maintenance, simplifying further the operation, avoiding the risks of human errors, and minimizing potential disturbances of the process. If we assume that the sludge can rise up to 90% of the reactor height (3.6 m in 4-m tall reactors), it is clear that only one annual sludge discharge would be necessary to dispose of the sludge grown during one year (about 2.4 m). After discharges, about 1.20 m of sludge would remain in the reactor. It can be calculated that 1.20 m of sludge would be enough to degrade (transform to methane) all the biodegradable organic matter retained in R1, assuming a sewage concentration of 400 mgCOD/L, a mean applied HRT of 6-8 h, a removal efficiency of about 70%, a sewage anaerobic biodegradability of 70%, and a sludge SMA at working temperature of 0.10 gCOD-CH_4/gVSS.d (with 25 gVSS/L of sludge). This calculation was made assuming that retained organic matter was immediately available for methanogenesis. As hydrolysis is known to be the rate-limiting step in the process of anaerobic digestion of particulate organic matter, it can be inferred that the excess SMA was even higher. After discharges, the concentration of VFA in the effluent should be monitored, in order to detect reactor overloading. To always keep an excess treatment capacity in the reactor, two or more discharges per year are recommended. Discharges should be performed during late summer months or at the beginning of autumn to ensure maximum sludge stability. If influent flow rate is very variable, sludge washout may be induced in peak hours, reducing the quality of the effluent. In this case, more frequent sludge discharges may be needed to keep the sludge always at an optimum level.

COD balances

COD balances for period IV (158 d), with the system assumed to be at steady state, are presented in **Figure 6**. More than half of the incoming COD was converted to methane in R1, while a relatively lower proportion was converted to methane in R2. Methane content in the biogas was remarkably constant in both reactors, and amounted to 90% in R1 and 95% in R2. This high methane content can be partially ascribed to the different solubility of methane and carbon dioxide, especially at low temperatures (Singh and Viraraghavan, 2003). COD_{tot} removal efficiency in R2 was rather low during this period (36.1%), and this is clearly reflected in the big size of term E.

On-site biogas measurements were not used eventually to calculate term G, as it turned out during the experiments that the gas collectors, due to a construction reason, could not reach the minimum flow rate and the pressure required by the gas meters to operate correctly. Instead, term G in the balances was calculated after the percentage of methanogenesis measured for raw sewage in bench-scale batch tests (see chapter 2). For that purpose, it was assumed that the degradation of the different COD fractions in the pilot plant was similar to that observed in laboratory tests. This assumption was justified because the SRT in the reactors was much longer than the average length of the laboratory tests and therefore, entrapped suspended solids (the bulk of the COD) and COD_{col} had enough time to be degraded up to the maximum possible extent. It was also assumed that biodegradable COD_{dis} was fully converted during its passage through the sludge bed. Percentages of methanogenesis used in the balances were 67.6 and 31.6% for R1 and R2, respectively. They were corrected to account for differences in temperature and the relative proportions of the different COD fractions between the sewage used in the lab and the influent to each reactor during period IV. Methane production in R2 may have been slightly overestimated with this method because the anaerobic biodegradability is expected to be lower in anaerobically pretreated sewage than in raw sewage.

The correction term C was found to be very low in both reactors when term G was calculated in this way. Average methane production during period IV was 0.26 and 0.12 $Nm^3/kgCOD_{removed}$ in R1 and R2, respectively, in agreement with values expected in full-scale UASB reactors (Haskoning and WAU, 1994).

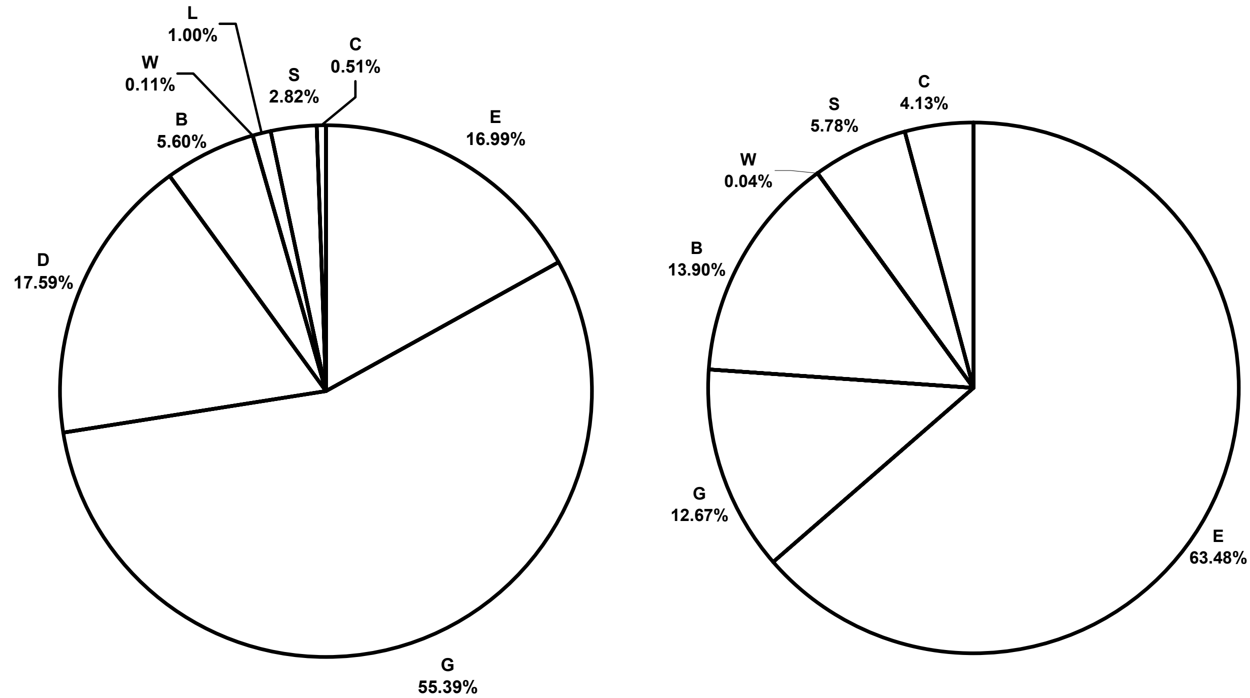

Figure 6. COD balances for period IV in R1 (left) and R2 (right). Slices represent the terms of the balance as percentage of influent COD. I: influent; E: effluent; G: gaseous methane; D: dissolved methane; B: sludge bed; W: sludge washout; L: scum layer; S: sulfate reduction; C: correction factor. There was neither scum layer formation nor additional methane dissolution in R2.

Sludge washed out from R1 was added to the influent COD (term I) in R2, proportionally to the influent flow rate.

In R2, term D is a net value, since dissolved methane from R1 enters into R2 with the influent. The concentration of dissolved methane in the influent of R2 was assumed to be in equilibrium with the methane concentration of biogas in R1. Diffusive stripping of dissolved methane between R1 and R2 was neglected, assuming that the methane concentration gradient in the gas phase was very small. This assumption was based on the following: (1) the effluent of R1 was always exposed to an atmosphere of biogas in the space around the GSL separator device on top of the reactor, in the pipes, and in the bucket C1 (see **Figure 1**); (2) temperature and pressure were constant; and (3) retention time in C1 was very short (about 1 min). Convective stripping from the top of the reactor and from C1 was also neglected, as long as these units remained always closed. It was calculated that a certain amount of dissolved methane left the liquid phase and was collected as gas in G2 since average liquid temperature in R2 was higher than in R1 during period IV. This amount could have been higher if effluent from R2 was supersaturated with methane (Batstone, 2000). Therefore, term *D* is zero and doesn't appear in the balance for R2 (**Figure 6**, right). It was also assumed that dissolved methane was not found as COD in laboratory analyses, because it left the liquid phase during sampling, storage, and measuring (more than 24 h exposed to the air).

During periods II and III, some sludge accumulated in the bottom of intermediate pumping containers and was incidentally observed in a couple of grab samples. Sludge was not detected in 24-h composite samples, probably because washout was not constant, while effective sampling time was very short (a few seconds every 3 h, only on sampling days). From experiments conducted with secondary settlers fed with the effluents from both reactors during period IV, it turned out that sludge washout was only a minor fraction of influent COD in both reactors. A distinction between solids from the sludge bed ("secondary" solids) and particles from the influent, which were not retained in the reactor ("primary" solids), is difficult to make. Intermittent washout (if observed) would be an indication that secondary solids are being lost, because short circuits over the sludge bed (main cause of primary solids washout) are assumed to be a constant phenomenon (van der Meer, 1979). The amount of scum layer withdrew from R1 was very low during the entire operation (see **Figure 6**) while no scum layer formation was observed in R2. COD in the scum layer from R1 was 56.8 ± 9.8 (n = 11) g/L of scum, similar to that of the sludge. Complete sulfate reduction was not verified in R1, in agreement with results presented by Haskoning (1996; 1996b) for UASB plants in Kanpur and Mirzapur (India).

Term C in the balances can be attributed to several causes, e.g.: (a) some parameters were averages over one entire year; (b) accurate determination of sludge bed height and scum layer volume was difficult; and (c) sludge and scum COD results were variable. COD balances built over R1 in period VI were similar to period IV, with term C lower than 1% (results not shown). Singh and Viraraghavan (1998) reported a COD gap of about 10 to 15% of the total input in COD balances performed over a UASB reactor treating sewage at 20°C, partially attributed to COD consumption for cell synthesis. Basic variables needed to build the COD balances were the following: flow rate, air and liquid temperature, atmospheric pressure, influent and effluent COD, biogas production and composition (or percentage methanogenesis for each COD fraction), sludge bed height, sludge COD, scum layer volume and COD, sludge washout, and sulfate concentration in influent and effluent.

Removal of fecal coliform

Removal of fecal coliform in the entire system during the first year of operation (periods I to IV) was 99.9999% (99.94% in the anaerobic steps and 99.98% in the WSPs). Final effluent concentration remained below 300 MPN/100 mL, complying with the World Health Organisation's recommended guideline for unrestricted irrigation (1000 MPN/100 mL) (WHO, 1989), and local discharge standards (2000 MPN/100 mL) (SeMADeS, 2001). Removal in WSPs approached that expected from design calculations, validating the hydraulic model of completely mixed tanks in series. It was reported that plug-flow regime is difficult to achieve in practice (Cavalcanti, 2003). Contrary to what is reported in literature (Elmitwalli, 2000; van Haandel and Lettinga, 1994), high fecal coliform removal was observed in the UASB reactors, although a clear reason for this could not be found. WSPs will provide new data to determine accurate kinetic constants for the design of full-scale systems.

FINAL DISCUSSION

Results presented in this study show that a drop in sewage temperature below 20ºC for several months can be easily assimilated by a UASB reactor without any deterioration of its performance, even when there is a high concentration of SS in the influent. This is in agreement with technical reports indicating that full-scale UASB reactors can cope with sewage temperatures around 18ºC in winter without a substantial reduction in their treatment efficiency (Haskoning, 1996; Haskoning, 1996b). It appears that, when sewage temperature is above 20ºC, UASB reactors can be safely used for sewage treatment at a design HRT of about 6-8 h. At temperatures below 20ºC, the percentage of hydrolysis of the retained SS needs to be known to predict the behavior of the system, and to calculate the HRT needed to reach a certain required SRT. It was reported that, under conditions prevailing in the middle east (strong sewage, winter sewage temperature of 15ºC), anaerobic sewage treatment can only be accomplished in single-stage UASB reactors when HRTs of more than 20 h are applied (Mahmoud, 2002). At this HRT, unacceptably big reactors would be needed, and complementary anaerobic or physical processes were proposed (Mahmoud, 2002). On the other hand, Halalsheh (2002) reported that most of the COD_{tot} in a two-step UASB system for sewage treatment in Jordan was retained in the first step, at an average working temperature of 18°C in winter and 25°C in summer, indicating that a second anaerobic step may not be indispensable under these conditions.

As described in chapter 2, Zeeman and Lettinga (1999) presented an equation to calculate the HRT needed in a UASB reactor to reach a certain SRT. **Table 7** shows HRT and V_{up} calculated with this equation for different SRTs, using percentages of hydrolysis (H) reported in literature for a sewage temperature of 15ºC. The parameters needed for the calculation (except H) were measured during the present study (rounded values were used for the sake of simplicity). According to Zeeman *et al.* (2001) and Halalsheh (2002), an SRT of 75 should be enough to provide methanogenic conditions and both hydrolysis and β-oxidation of lipids at 15ºC. Singh and Viraraghavan (2003) reported that the minimum SRT needed to operate UASB reactors treating municipal wastewater at temperatures above 6°C was between 35 and 55 d. Under the conditions of this work, an SRT of 75 d could be reached by applying HRTs around 10-12 h in a single-stage UASB reactor (see **Table 7**). This suggests that additional treatment steps may not be needed even when mean sewage temperature is 15°C. A reactor designed with an HRT of 12 h would be twice as big as one designed with an HRT of 6 h. However, the operation of a single-stage UASB reactor is much more simple than the operation of a two-stage system, or a UASB reactor with an additional (heated) sludge digester. Operation and maintenance costs would also be much cheaper for a single-stage system. This is an important aspect in developing countries, where initial investment costs for sewage works are generally covered by international loans, but local governments must afford the running costs. It is even likely that the construction of complementary units outpaces the cost of additional volume in a single-stage UASB system.

The difference between average annual temperature and winter temperature in sewage during this study was 3.6ºC. If this difference is assumed to be constant for medium-size cities in subtropical regions with similar social and cultural conditions, and 15°C is considered as the minimum acceptable temperature, it can be concluded that anaerobic sewage treatment could be applied in single-stage UASB reactors in places in which mean annual sewage temperature is higher than 18.6ºC. Unfortunately, sewage temperature measurements are generally performed only at sewage treatment plants. Therefore, there are normally no data available before treatment plants are constructed and put into operation. But ambient temperature is generally recorded as a routine procedure by governmental agencies, airports, and different types of private companies. In Salta, sewage temperature was 6.5ºC higher than ambient temperature. Therefore, if the minimum required average sewage temperature is 18.6ºC, the minimum average ambient temperature needed would be 12.1ºC. In subtropical regions, there is usually a well-defined cold period of only 2 to 4 months. Average (monthly) temperatures close to 15ºC are only expected during this period. Therefore, anaerobic sewage treatment in single-stage UASB reactors could be safely recommended for subtropical regions as a whole. This single consideration would represent a substantial increase in the number of cities and towns around the world that may now “qualify” for anaerobic sewage treatment! Additional treatment steps may have to be considered only if mean (monthly) sewage temperatures below 15ºC are common during the coldest period of the year.

High removal efficiencies observed in R1 can be explained by the fact that secondary sludge present in concentrated sewage was probably fully retained in this reactor as, by definition, secondary sludge can be settled in 2 h. The performance of the reactor when it is fed with “normal” raw sewage can be estimated by subtracting from concentrated sewage the amount of extra COD incorporated by secondary sludge.

The characteristics of the sewage from a particular community can be explained, to a great extent, by the social behavior of its members. Therefore, the relationship between the different COD fractions of sewage is likely to be rather constant from year to year. The original concentration of raw sewage during the period in which secondary sludge was recirculated can be calculated back if the original proportions of COD_{sus}, COD_{col}, and COD_{dis} are known for raw sewage. During one year of measurements prior to sludge recirculation, the ratio $(COD_{col}+COD_{dis})/COD_{sus}$ in raw sewage from the city of Salta was 0.77 (n = 183). As secondary sludge is composed mainly of SS, it can be assumed that COD_{col} and COD_{dis} in concentrated sewage came only from raw sewage. In fact, COD_{col} and COD_{dis} were similar in both raw and concentrated sewage during about 3 months of parallel measurements (see chapter 2). Under these assumptions, the original concentration of COD_{sus} in raw sewage was obtained as $(COD_{col}+COD_{dis})/0.77$. The original COD_{tot} in raw sewage would then be (calculated) COD_{sus} + (measured) COD_{col} + (measured) COD_{dis}. In periods IV and VI, when R1 was assumed to be at steady state, removal efficiencies in R1 based on this “calculated” raw sewage were lower than those obtained with concentrated sewage (**Table 8**).

However, these efficiencies were more comparable to values reported in tropical countries under similar hydraulic conditions (see chapter 1). Site-specific studies should be performed to assess the feasibility of anaerobic treatment in particular cases, especially when sewage temperature can be a limiting factor, or sewage composition detaches too much from that observed in this work. Studies must include the determination of the following parameters, needed to estimate the required HRT in the anaerobic reactor: (a) sewage temperature; (b) sewage composition, including the determination of the different COD fractions; and (c) sewage biodegradability and hydrolysis under average and minimum local temperature.

A UASB reactor serves both as a biological reactor and a settling tank. The operation of a UASB reactor as a settling device is as important as its biological function, especially in the treatment of domestic wastewater. However, the physical events taking place in a UASB reactor are quite complicated. Several design parameters will influence the retention of SS in the reactor, among them the upflow velocity, the sludge bed height, and the rate of biogas production (Haskoning and WAU, 1994). The physical behavior of the SS in the reactor and the amount and composition of the sludge that will develop are crucial for the estimation of the reactor efficiency. These parameters can only be determined in lab or pilot-scale experiments with UASB reactors, although values reported in literature can be used as a first approximation.

Table 7. HRT and V_{up} calculated with the equation proposed by Zeeman and Lettinga (1999) as a function of SRT for different percentages of hydrolysis (H) reported in literature for a sewage temperature of 15°C. C = influent concentration; SS = particulate fraction of COD influent; X = expected sludge concentration in the reactor; R = fraction of the particulate COD which is expected to be removed.

Parameters[a]	H = 30.0% (Zeeman *et al.*, 2001)			H = 15.0% (Mahmoud, 2002)			H = 24.3% (Halalsheh, 2002)		
SRT (d)	**50**	**75**	**100**	**50**	**75**	**100**	**50**	**75**	**100**
HRT (h)	6.7	10.1	13.4	8.2	12.2	16.3	7.3	10.9	14.5
V_{up} (m/h)	0.60	0.40	0.30	0.49	0.33	0.25	0.55	0.37	0.28

[a]Parameters used in the calculation were obtained during this study:
C = 0.4 gCOD/L; SS = 0.75; X = 30 gCOD/$L_{reactor}$; R = 0.80.

Table 8. Concentration and removal efficiency in R1 during periods IV and VI for concentrated sewage and calculated raw sewage.

Period	COD	Influent (mgCOD/L) Concentrated sewage	Influent (mgCOD/L) Calculated raw sewage	Effluent (mgCOD/L)	Removal efficiency (%) Concentrated sewage	Removal efficiency (%) Calculated raw sewage
IV	COD_{tot}	472.5	208.3	80.3	83.0	61.5
	COD_{sus}	360.1	108.5	27.7	92.3	74.5
	COD_{col}	37.9	37.9	10.4	72.6	72.6
	COD_{dis}	45.2	45.2	34.9	22.8	22.8
VI	COD_{tot}	362.9	265.2	94.6	73.9	64.3
	COD_{sus}	282.1	138.2	41.3	85.4	70.1
	COD_{col}	46.0	46.0	21.4	53.3	53.3
	COD_{dis}	59.8	59.8	33.8	43.4	43.4

CONCLUSIONS

- The pilot-scale sewage treatment system studied was highly efficient under subtropical conditions (mean sewage temperature = 23.0ºC). The final effluent consistently complied with discharge standards for COD and fecal coliform.
- The startup of the first UASB reactor was successfully achieved in one month, without inoculum, at an HRT of 8.2 h (V_{up} = 0.48 m/h).
- A COD_{tot} removal efficiency of 89.2% was obtained in the two-step anaerobic system at HRTs of 6.4 + 5.6 h (V_{up} = 0.62 + 0.70 m/h), with 83.0 and 36.1% removal in the first and second steps, respectively.
- COD removal efficiency achieved in the first UASB reactor at steady state was similar to that reported in full-scale UASB reactors in tropical countries operating under similar hydraulic conditions.
- Fecal coliform removal in the whole system was 99.9999% (99.94% in the anaerobic steps and 99.98% in the WSPs).
- The performance of the system was not affected during the coldest period of the year, which lasted more than three months. During this period, mean sewage temperature was 19.4ºC, minimum monthly temperature was 17.2ºC, and minimum daily temperature was 12.6ºC.

- The use of coupled COD balances allowed a better understanding of the process, and was useful to check the accuracy and consistency of the measurements. A minimum set of data needed to build comprehensive COD balances over UASB reactors is proposed.
- SMA in the first and second UASB reactors was 0.12 and 0.04 gCOD-CH_4/gVSS.d, respectively.
- Methanogenic activity measured in different anaerobic sludges was significantly higher at 30ºC than at 20ºC. Methanogenic activity at 20ºC was 65% of the activity at 30ºC.
- The substrate concentration needed to obtain the maximum possible methanogenic activity in SMA tests should be previously established for each type of sludge.
- The sludge growth rate in the first UASB reactor was low, and only one or two discharges per year could be enough to dispose of the excess sludge. Discharges should be performed in summer, when the sludge was found to be fully stabilized. VS reduction in all anaerobic sludges complied with the EPA standard for sludge stability.
- Under the conditions of this work, the second anaerobic step was not necessary to achieve high COD removal efficiency, a good quality effluent, and satisfactory sludge stabilization. One UASB reactor followed by WSPs could be a very efficient system for sewage treatment in subtropical regions.

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CHAPTER 6

Assessment of the sustainability of anaerobic sewage treatment in Salta, Argentina

ABSTRACT

The basic contention of this chapter is that the concept of "sustainability" should be used as a criterion for the assessment of (environmental) technology. The need for a rational, simple, and comprehensive method to assess the sustainability of different technologies is emphasized and discussed, based on the idea that there are none or only a few technologies that can be considered intrinsically sustainable, independently of the context in which they are used. The sustainability assessment should take into account the global and long-term effects of the use of a particular technology. As an example, the sustainability of different sewage treatment technologies was assessed for the city of Salta, located in a subtropical region in northwestern Argentina. Three types of systems were compared: (A) aerobic high-rate treatment system (primary and secondary settling, trickling filters, sludge digestion, and chlorination); (B) waste stabilization ponds (WSP); and (C) upflow anaerobic sludge bed (UASB) reactors with posttreatment in polishing ponds (a type of WSP). The selection of these technologies was based on their immediate availability in the region. Technologies were compared according to 4 basic criteria subdivided into 20 operational indicators. Calculations were performed as in weighted-scale checklists and results were presented in a "sustainability matrix". Criteria and indicators were first weighed according to their relative contribution to the local environmental and social system (Importance). After that, the different options were assigned a score with respect to each indicator (Performance). For each technology, a so-called sustainability index (SI) was calculated, as the sum of the products between importance and performance for all the indicators. According to the SI calculated and a predefined (arbitrary) scale, sustainability was medium for options A and B, and high for option C. The method used was simple and sensitive to detect differences in sustainability between the options compared. Validation by a representative group of local stakeholders is required before policy decisions are actually made. A fully democratic way of decision-making that can go beyond political and economic motivations and that may be able to acknowledge and solve environmental problems and social injustices is probably the only way to achieve sustainable development.

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INTRODUCTION

The technical performance of anaerobic reactors for sewage treatment under local conditions has been extensively discussed in chapters 2 to 5. It was demonstrated, beyond any reasonable doubt, that the systems studied were efficient and reliable, and that their full-scale use was highly feasible in the region. However, local companies, engineers, and decision-makers remained rather skeptical about the potential use of anaerobic technology, even though the technical conclusions attained in our studies were not disputed. One of the main concerns was the lack of experience with anaerobic systems in the region. At the same time, existing systems like waste stabilization ponds or aerobic trickling filters were viewed as similarly efficient and comparable in terms of costs. Moreover, other more subjective issues were also raised, like the potentially low acceptability of a new technology or concept by local sanitary engineers and companies, and the probable rejection that this technology would cause on end-users on fears of failures, bad odors, and risks of poor maintenance. These types of arguments were completely unfounded and very frustrating but, at the same time, revealed that a purely scientific, technical, or even economic assessment would not guarantee the acceptance and implementation of a successful technology. It was clear at that point that the assessment of the different technological options available should be broadened in order to include criteria other than only technical features and costs. A comprehensive assessment tool seemed to be needed to show the advantages and disadvantages of a set of potentially available technologies based on an alternative assessment criterion acceptable and understandable to all parties involved.

In this chapter, it will be argued that “sustainability” can be the alternative and comprehensive criterion to assess technologies, systems, concepts, or policies. The use of this criterion as a decision-making tool is highly needed on practical grounds, as discussed above, but has also profound theoretical and moral roots. The concept of sustainability, or “sustainable development” will be discussed, highlighting the different approaches proposed in the literature.

Environmental technology will then be presented as one of the elements that can help countries (both developed and developing) to follow the path of sustainability, provided that the selection of a particular technology is based on a rational and democratic assessment of its contribution to sustainability in the local and global context. The usefulness of some of the methods proposed in literature will be briefly discussed.

The chapter finishes with an example (or introductory case study) in which the sustainability of three technological options for sewage treatment are assessed for the city of Salta (Argentina), using a weighted-scale “sustainability matrix”. The method used in this example should not be regarded as a blueprint about how to do a sustainability assessment. Each case requires particular attention and the methodological approaches may vary. The most important message of this work is that the different (technological) options available for a specific purpose, in a given location, at a certain time, *should* be assessed on the basis of their contribution to local and global, short-term and long-term sustainability. Different techniques may be used, provided that the basic assumptions behind these techniques, and their advantages and disadvantages, are clear for all involved in the assessment. Transparency in the selection of the method, in the process of assessment, and in the final decision making, together with a representative democratic participation of all stakeholders can be regarded as an effective “protection” against the disadvantages of any of the methods.

The main objective of this study is to connect the issue of sustainability with the selection, transfer, and adoption of (environmental) technology. It is considered important that scientists and technologists from different backgrounds take active part in the debate around the issue of sustainability, especially when technological decisions have to be taken.

SUSTAINABLE DEVELOPMENT

The concept

The World Commission on Environment and Development (WCED) chaired by the then Norwegian prime minister, Gro Harlem Brundtland, defined sustainable development as “development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs” (WCED, 1987). The term “sustainable development” was apparently first used by the International Institute for Environment and Development (IIED) based in London (Mebratu, 1998), although Lester Brown, founder of the Worldwatch Institute, is said to be the coiner of the term (Thompson, 2001). The origin of the idea is also related to “The World Conservation Strategy”, a document produced by the International Union for the Conservation of Nature (IUCN) in 1980 (Holland, 2003). However, it was the report by the WCED (also called the “Brundtland report”) which put the term on the international agenda (McNeill, 2000). One of the main precursors of the concept of sustainable development was the idea of what was later called “appropriate technology” advocated by Ernest F. Schumacher in his historic book *Small is Beautiful* (1979).

Different views

The WCED definition of sustainable development was considered by many as vague and ambiguous, although these alleged features have probably contributed to its worldwide acceptance (Mebratu, 1998). Mitcham (1995) presented a discussion on the origins and ambivalence of the concept of sustainable development, which was seen, nonetheless, as a potential tool to bridge the gap between the so called no-growth and pro-growth factions, the “misers” and the “fraudsters” in the words of Huber (1991). Tijmes and Luijf (1995), on the other hand, challenged the very foundations of the concept, as it has been articulated by the WCED, arguing that it only extends the principles of the economics of scarcity into the environmental debate. In their work, they point out that the WCED adheres to the economic idea that, in modern market-oriented societies, economic growth will always be necessary to contain the increasing rivalry between individuals in their never-ending competition for scarce goods. On the contrary, in traditional cultures rooted in religious and solidarity principles, needs were not seen as endless or indefinite, but as religiously and culturally constrained. They conclude by saying that qualifying growth and development as (potentially) sustainable is simply a contemporary attempt to hide the ambivalence between the concepts of economic growth and scarcity. Many alternative definitions of the concept of sustainable development have been introduced, in an attempt to make the concept more acceptable, more precise, or more operational (Mitcham, 1995; Mebratu, 1998; George, 1999; Rijsberman and van de Ven, 2000; McNeill, 2000).

Two main streams can be differentiated in the sustainability debate, in line with previous divisions in the environmental debate. These two approaches have been broadly defined as *technocentrism* and *ecocentrism* although variations within and combinations between them have been identified (Pepper, 1996; Mason, 1999; Lee *et al.*, 2000).

Technocentrism acknowledges the existence of environmental problems but relies on technological and managerial solutions working within the framework of the market economy to solve them (Mason, 1999). An example of technocentrism is the theory of “ecological modernization”, which is described as a tool to analyze and predict processes of transformation in industrial societies which could allegedly help to anticipate, understand, push, and guide future development (Spaargaren and Mol, 1992; Mol, 1995). Ecological modernization advocates that there is a possibility of overcoming the environmental crisis without radically changing the present patterns of development because, for example, “there is no reason to assume an automatic and unbreakable link between economic growth and increased energy consumption” (Weale, 1992). Some of the changes observed in the Dutch chemical industry, for instance, were explained through the theory of ecological modernization (Mol, 1995). More examples of sectors moving in this direction, not only in industrialized countries but also in developing countries, were foreseen by Pachauri *et al.* (1995). Mol (1995) predicts that, in a globalized world, if the concept of sustainable development (with ecological modernization as its operational tool) is adopted in industrialized countries, it is likely that it is also going to be adopted worldwide, as it happened with the concept of industrialization in the past.

Ecocentrism, on the other hand, is skeptical of large-scale technological developments and the commitment of corporations to environmental matters, and raises ethical issues as the main driving force for the protection of nature (Mason, 1999). In this context, the concept of sustainable development, when it is labeled as “ecological modernization”, “eco-liberalism”, or simply “global development” is regarded as just another product of the market economy that could never cure the crisis that the very market economy has helped to produce. Under this idea, several approaches to sustainability have been proposed, involving more political and economic decentralization (Lee *et al.*, 2000). The empowerment of local communities, the importance of cultural differences, and the right of every nation to effectively own its environmental assets lie at the basis of some of these approaches (Leff, 2000). As pointed out by George (1999), we should not fail to differentiate development that is not equitable but is artificially sustained for long periods of time from real long-term sustainable development in which social justice is a fundamental component. The equitable access to environmental resources (and the egalitarian sharing of collateral burdens) should be as important to reach sustainability as the overall throughput of the economy (Valentin and Spangenberg, 2000). The notion of sustainable development is moving from a more environmentally oriented idea to a more socially directed concept. For now, some environmentalists feel compelled to specify that they are not only pursuing an environmentally friendly world, but also a socially just one (Worldwatch Institute, 2003).

Mason (1999) introduced the concept of “environmental democracy” as a comprehensive synthesis of environmental and social concerns, away (or, better, equidistant) from both technocentrism and ecocentrism. This concept is a challenge to the usefulness of market-based or bureaucratic decision-making models “to generate decisions which are responsive to social and ecological concerns” and an attempt to neutralize the lack of social thinking in some conventional environmentalism. In his words,

> "…*civic self-determination, as opposed to, say, elite political bargaining or market choices, is the most legitimate means of generalizing environmental interests in a democratic fashion*".

Weale (1992), in line with this idea, stated that

> *"Protecting the environment from pollution has now become one of the central issues in the politics of industrial and post-industrial societies. Acid rain, global climate change with its threat of greenhouse warming, the dispersal of chemicals in the environment, keeping waters clean for drinking and recreational purposes, and the national and international management of toxic wastes have ceased to be the concerns of a few specialists or committed ecologists and instead are widely discussed and argued about in the press and the media, among political parties, between citizens and their representatives, or among governments in international negotiations. Ensuring that affluent societies do not transgress the boundaries imposed upon their production and consumption by the renewing and cleansing cycles of the ecosphere, or 'making peace with the planet' as Barry Commoner eloquently expresses it, is increasingly likely to be one of the means by which democratic governments secure their legitimacy and their right to exercise power*".

A fully democratic way of decision-making that can go beyond merely political and economic motivations and that may be able to acknowledge and solve environmental problems and social injustices is probably the only way to achieve sustainable development. This conceptual framework supports the overall idea behind this chapter.

Sustainable development implies and demands that all basics needs belonging to the physiological and social level (food, housing, health, education, security, human rights, equity, dignity, freedom, etc.) are fully guaranteed. In this context, the basic goal of all sustainable development policies should be to "ensure a healthy environment and a just civil society for everybody, everywhere" (Neefjes, 2000), and forever. There have also been attempts to incorporate spiritual issues as a fundamental part of the health and well being of a given society (Chuengsatiansup, 2003). A number of factors enhancing spiritual life (grouped as "supportive infrastructure" and "conducive environments") and paradigmatic changes detrimental to spiritual health have been identified. Therefore, there may be ways of promoting, protecting, or nurturing spiritual aspects taking the appropriate measures and following adequate policies. As Wilber (1998) eloquently put it, the basic problem of western modern societies is that they have lost sight of the interior (spiritual) dimensions of the world, leaving people in a "flatland" devoid of meaning and value. He defined "transcendental naturalism" as the operational framework within which modern knowledge and pre-modern wisdom can reach a much needed reconciliation, without which "the future of humanity is, at best, precarious".

Realization of sustainability

Despite its limitations and criticisms, the concept of sustainable development is recognized to be the first institutionalized attempt to balance the three dimensions of development: economic welfare, environmental protection, and social justice (McNeill, 2000). Institutional aspects are necessary means to achieve sustainability and have been proposed as a fourth (operational) dimension (Valentin and Spangenberg, 2000). The notion of sustainability provides a new framework within which issues of growth and environment can be discussed (Holland, 2003).

The pursuit of sustainable development will require changes in domestic and international policies, irrespective of differences in economic and social systems and ecological conditions among countries (WCED, 1987). As Gatze Lettinga put it, "… what sense does it make to pursue a paradisiacal natural environment in a single country or region, when at the same time little if any money or technology is made available to contribute to highly needed environmental improvement in less prosperous countries?" (Lettinga *et al.*, 2001). The very notion of sustainable development implies a never-ending effort to establish proper life conditions for present and future generations. This effort will enable people of all times to fully develop their talents putting them in a better position to contribute to the further improvement of the social structure and the quality of the environment. Sustainable policies must not be limited to a specific country, region, or group, and must ensure conservation and/or optimal exploitation of all natural resources recognizing that human beings are the most valuable of all resources available.

Most of the enormous technological developments achieved in the context of western societies originate from the willingness of people (nations) to maintain and improve their material prosperity. To a much lower extent these developments originate from the desire to improve the well being of all people on earth. There seems to be a general agreement about the intrinsic goodness of concepts like "progress" and "growth". However, none of the leaders of this time seems to know exactly where this road will eventually lead us. The mechanisms of social and economic development in the world are difficult to manage, and it looks like they are also poorly understood. On the other hand, many technological innovations, meaningless or even negative in the light of sustainable development, continue to find their way to implementation pushed only by short-term economical interests.

The acute process of centralization of means and power and the colossal economies of scale that accompany the so-called globalization of the world may be sometimes economically "efficient" but they are also increasingly vulnerable. Besides, this process tends to widen the gap between the rich and the poor, the developed and the underdeveloped. Even in the prosperous parts of world, social-economic developments are not analyzed at a macro scale, from a long-term perspective, and in a more holistic setting. However, a positive side of the process of globalization is that it also implies a wider dissemination of consciousness about the situation in the world that can, in turn, promote sustainable "bottom-up" and "top-down" decentralized initiatives. In this context, it is important to view sustainable development more as a direction to follow, a road, a pathway, and a constructive utopia, than a specific goal. Interestingly, if sustainability is defined as a road and not as an end point, the moment we take this road we are, by definition, in a sustainable situation. We don't know exactly where we go, but we know quite well where we don't want to go. When we take this road we are in, say, point A and we want to go in the direction of point B (sustainability). We don't know exactly where B is, but we know that we don't want to go in the direction of C, D, and so on. In that sense, the process of going from A to B should not be called a "transition", because it's not going to be a temporary endeavor, but an "evolution", because it will be a permanent process. Nobody can be sure now about how a sustainable society would look like. However, we know beyond any reasonable doubt that some activities, if not stopped or changed, will certainly not contribute to it. In that sense, we could and should start to take decisions based on the principles of sustainability, basically a search for justice in space and time.

In normal practice, steps in the direction of more sustainability must be small because new developments have to fit into an existing frame inherited from the past. Proceeding in bigger steps might have such an impact on society that they can not be smoothly accommodated, e.g. they might imply a destruction of previously invested capital or might greatly exceed the economical/financial means of the local communities involved. There is no reason to promote changes that a community, city, or region can not adjust to (for economic, cultural, or social reasons), but the main point is that things must be developed in the proper direction, in the direction of sustainability. The consequence of this is that all new investments, changes, innovations, policies, and technologies need to be carefully assessed on the basis of long-term sustainability criteria. In the specific field of sanitation, more sustainable options should be immediately started, adopting a step-wise approach, especially when there is a need to change or convert existing conventional systems.

Technology diffusion and transfer

Technology, understood not only as "hardware" (artifacts) but also as "software" (skills needed to develop, produce, and use artifacts) (Weaver, 2000), is found throughout all institutions and sectors of society, and is one of the most important factors for development (Freeman, 1974). The developed world, with less than one third of the world's population, has more than 93% of the world's scientific and technological capabilities, 65% of material resources for development of science and technology, and 99% of scientific and technological information (Menghistu, 1988). However, sometimes the modern and efficient technologies created within the context of market economies have been associated with heavy social and ecological costs (Bandyopadhyay and Shiva, 1989). A discussion on different definitions of technology can be found in Weaver (2000).

For the purpose of this work, "environmental technology" can be defined as the branch of technology dealing with the detection, study, and solution of real or potential problems affecting the natural equilibrium. A large variety of terms have been used to refer to different types of environmental technology, initially divided into *end-of-pipe* technology (corrective technology added to polluting processes) and *clean* technology (the processes themselves generate less wastes and pollution) (OECD, 1985). The term *cleaner* technology is sometimes used, defined as all techniques, processes, and products that avoid or diminish environmental damage and/or the usage of raw materials, natural resources, and energy (Weale, 1992). Skea (2000) (based on ACOST, 1992) classified environmental technology into seven categories: pollution control, waste management, recycling, waste minimization, clean technology, measurement and monitoring, and clean products. Experience in some industrialized nations has proven that, in many cases, the use of specific environmental technologies offers both environmental and economic advantages (Weale, 1992; Skea, 2000). Although it can only be proven in a limited number of cases, the potentially win-win character of some environmental policies and regulations is also known as the "Porter Hypothesis" (Gabel and Sinclair-Desgagné, 1998).

Many environmental and pollution problems have international dimensions (Folmer and de Zeeuw, 2000). However, considerable amounts of environmental policies have been produced in the framework of conventional and domestic policy-making (Porter and Brown, 1991). At the same time, as environmental problems are common to all countries, policy "borrowing" in these issues has also been attempted, especially between industrialized countries (Weale, 1992). Environmental problems aggravated (at least locally) the situation of poverty and hunger in many developing countries (Worldwatch Institute, 2003), although there is still some controversy about how serious environmental problems really are worldwide (Sagoff, 2000; Lomborg, 2001). By applying an anticipatory policy, rather than a reactive one, substantial savings can be made because clean-up costs are generally greater than prevention costs (Weale *et al.*, 1991). The technology needed to implement a borrowed environmental policy may not be the same as that employed in the country of origin. Use of local materials, different manpower input, and many other variations are possible, provided that the scope of the policy is maintained. The phase of implementation is decisive for the successful adoption of any policy or technology, and not only when it is borrowed from another country (Vogel, 1986).

The introduction of new technologies can transform socio-cultural systems and therefore, a certain process usually needs to take place between the invention or development of a certain technology and its effective adoption by end-users (Freeman, 1974). The diffusion of new or improved technologies in society is a complex process, facilitated or hindered by a large number of factors (Ray, 1984). The evolution and adoption of a new technology has been explained with the invention – innovation (introduction) – diffusion – decline model (Rosenberg, 1976; Schot, 1991). It is generally accepted that the evolution of technology is steady and gradual and that many inventions could be explained as a logical continuation of the technology currently at use at that time (Basalla, 1988; Kemp, 1993). This evolution was not random, but has been generally fostered by conscious efforts in research and development (Weaver, 2000). The patent system has probably been beneficial in promoting inventiveness and localized economic growth under certain circumstances but the positive or negative effects of the patent system on the economy as a whole are controversial, including its potential for the creation of monopolies (Basalla, 1988). Four modes of technological innovation have been proposed, namely *incremental innovations* (small-scale modifications to existing technologies), *radical innovations* (discontinuous events of technological change), transformations of a *technology system* (affect several branches of the economy and may give rise to entirely new sectors), and changes in the *techno-economic paradigm* (affect the "style" of production and management throughout the economic system). Each mode is a step up to a higher level of complexity and social significance (Hellström, 2003).

Technology transfer allows developing countries to use technologies developed elsewhere, without being involved in the long and costly process of technological creation. The importer is supposed to save time and money although certain aspects of technology transfer has other social, economic, and technological aspects that have to be considered (Wei, 1995). Technology transfer should not only be a transfer of capital goods and operating skills and tools, but should also represent a base for developing the technological capability of a country. Building technological capability means the development of human resources necessary to select, assimilate, adapt, improve, and create new technology (Menghitsu, 1988; Putranto *et al.*, 2003). There has been a shift in the approach of international funding agencies towards supporting the construction of more research and technological capabilities in the South (Vessuri, 2003). According to Trindade (1994), technology transfer can only be successful when the following conditions are met in the receiving country: (a) a favorable policy climate; (b) availability of specific institutions; (c) a certain degree of basic education; and (d) the presence of domestic and international research partnerships to overcome critical mass limitations without deepening current patterns of technological dependency. Only in this case, developing countries will have the possibility of enjoying the relative "advantage of late-comers" (Pachauri *et al.*, 1994) and avoid mistakes incurred in the past, when inadequate borrowing of western technology, while producing some impressive results, also increased poverty and accelerated environmental degradation (Chamala, 1990).

Technological progress, while fostering an improvement in the quality of life, exposes us to previously unknown and some times catastrophic risks (Sinclair-Desgagné and Vachon, 2000). Any technology transfer will also entail the adoption of the risks associated with a particular technology, and an integrated assessment of innovation and risk is a prerequisite for a responsible form of technological innovation (Hellström, 2003). The amount of risks associated with new technologies (like vulnerability related to increased dependency on worldwide information networks) is usually related to the fact that most technological innovations tend to increase the amount of complexity, interdependence, and centralization in the world, relying more and more on a small number of corporate actors, and large economies of scale. Therefore, it can be argued that technological changes in the direction of simplicity, independence, self-sufficiency, and decentralization would entail an associated reduction in these global risks.

The speed at which a new technology will become adopted is affected by many technical, social, institutional, and even geographic factors (Schot, 1991). The adoption of (environmental) technology may also clash against established technological traditions, patterns, standards, archetypes, or models (in general called "paradigms") which are rooted in society and in scientific and technological research. A technological paradigm defines the needs that are meant to be fulfilled, the scientific principles utilized for the search task, and the technology to be used (Kemp, 1993). The path of technical change that develop from a technological paradigm is called a "technological trajectory" (Dosi, 1988, according to Kemp, 1993). In addition, reluctant attitudes among established groups working in conventional technologies frequently make the adoption significantly slower. In this respect, MacKenzie and Wajcman (1999) describe (and criticize) two contrasting visions: the deterministic vision (also called neoclassical), which basically states that the best technology (in terms of intrinsic technical efficiency) will eventually prevail, and the paranoid vision that sees all unsuccessful technologies as victims of a monopolistic complot against them. According to them, both sides "underestimate the complexity and uncertainty of knowledge of the characteristics of technologies, even the most 'technical' characteristics". In real cases it might be necessary to "weigh up the relative importance of differing characteristics" before selecting the best technological option for a particular situation.

Sustainability assessment

The question of the sustainability of the use of (all types of) technology is becoming more and more important. The assessment of the sustainability of a system, concept, or technology should attempt to consider the widest possible scope of impacts in society and the environment (in space and time) of the adoption and use of this system, concept, or technology. Local cultural factors have to be taken into account, especially in developing countries (Wad and Radnor, 1984). Attention should also be given, whenever possible, to long-term aspects at regional, continental, and global scale.

Since the assessment needs to be holistic, it seems impossible to devise a unique methodology applicable everywhere (Shrader-Frechette, 1985). Additional efforts must be done by assessment practitioners to adapt current methodologies and make sure to include specific sustainable development criteria in the analysis, while attempting a fuller integration of environmental, social, and economic aspects (George, 1999). The assessment must be rational, democratic, and explicit, and results need to be unambiguous, specific, and concrete no matter how controversial it might be to assign relative weights to the different alternatives (Lindholm and Nordeide, 2000). Results must be understandable by policy makers and the public at large, and provide some sort of qualitative or quantitative measure of the (comparative) sustainability of various technological options. If such a methodological effort is not made, the final decision on the adoption (or not) of any new technology (sustainable or not) will continue to be arbitrary, based only on the intuition, subjectivity, or "wisdom" of generally uninformed (and sometimes corrupt) policy makers and politicians. Indeed, daily practice shows that decisions and policies are mainly dictated by short-term interests in a quite "tunneled" way. The lack of appropriate parameters to quantify and assess the sustainability of different systems or technologies can be seen as one of the reasons why people, companies, institutions, and governments delay the adoption of more sustainable solutions (Lee *et al.*, 2000; Lettinga *et al.*, 2001). A strong defense of a systematic, rational, and democratic societal decision making was presented by Shrader-Frechette (1985).

No matter what kind of sustainability assessment method is used, it has to be considered that there are few systems, concepts, or technologies that are intrinsically sustainable, as there are generally no purely objective answers to the complex issues of sustainability. It can be argued that the minimum unit that may or may not be sustainable is the *scenario* in which technologies or systems are used and applied, rather than the technologies or systems themselves (and always in a global, long-term context). In this respect, simple dichotomies like, for instance, small-scale versus large-scale technologies, or centralized versus de-centralized systems loose their meaning, because criteria like "scale" or "degree of centralization" can no longer be used in isolation to assess the sustainability of a given technology, system, or concept (van Vliet, 2004). Moreover, different places may require different solutions. The successful implementation of the "best" solution for a particular case will depend to a great extent on the conscious, rational, unbiased, and democratic acceptance of local stakeholders (Rijsberman and van de Ven, 2000).

Assessment methods

A wide variety of tools have been used to assess the sustainability of a given technology, e.g. technology assessment (TA), environmental impact assessment (EIA), social impact assessment, land evaluation techniques, integrated resource management, multi-criteria analysis, among others (Dalal-Clayton, 1993). Modified EIA practices oriented to solve specific problems were seen as the most convenient method to assess the sustainability of sanitary projects (Lindholm and Nordeide, 2000). These practices are also supported by environmental advisers working in bilateral and multilateral development agencies (Dalal-Clayton, 1993). Life cycle analysis (LCA), a well structured and standardized method that enables comparison of results from different studies, was also used to assess the sustainability of different wastewater treatment options (Balkema *et al.*, 1998; Balkema, 2003). Balkema *et al.* (2002) discussed the application of four methods for the assessment of the sustainability of wastewater treatment systems: exergy analysis, economic analysis, LCA, and system analysis. Based on experiences in Norway, Lindholm and Nordeide (2000) concluded that LCA was too complex to be effectively used by standard sanitary engineers working in municipalities. The applicability of LCA in developing countries seems even more debatable, because most of the large quantity of data needed to perform it is generally lacking. Besides, other techniques like the so-called emergy analysis, may be more convenient than LCA in specific cases (El-Mashad, 2003).

The role of Cost-Benefit Analysis (CBA)

CBA is a tool for policy and project analysis designed to measure the net contribution of a project or public policy to the economic well being of the members of society (Hanley, 2000; Freeman III, 2003). Its use for the assessment of environmental impacts related to projects and policies is strongly recommended by many economists. This recommendation is primarily based on its two defining characteristics, namely: (a) it is based on the so-called "economic efficiency"; and (b) it is backed up by the theory of "economic welfare" (Hanley, 2000). The fundamental assumptions behind CBA could be summarized as follows:

a) The aim of economic activity is to increase the well being or economic welfare of individuals, who are the best and only judges of how well off they are in a given situation.
b) Economic efficiency is the basic (and only) criterion to assess welfare. A project or policy is basically efficient, in economic terms, if the benefits outnumber the costs. In this context, benefits and costs are not only purely economic, but also social and environmental. An economy is said to be efficient when no one can be made better off without making someone worse off (independently of the individuals' absolute economic situation).
c) Everything can (and must) be expressed in economic units (basically money), in order for the comparison of costs and benefits to be meaningful. This translation, when it comes to environmental "goods and services" is done via the method of "valuation". Eventually, valuation relies on the subjective opinion of potential consumers of the good or service, who are assumed to know all positive or negative effects of environmental amenities on their welfare (Shechter, 2000).
d) When expressed in terms of money, goods and services are assumed to have the property of "substitutability". This property implies that a good or service can be substituted by other of the same value, or eventually sold (or even destroyed) as long as the proper prize is paid in exchange. In other words, "trade-offs" between goods of different nature are allowed.
e) When costs and benefits have to be assessed over time, they have to be discounted at an appropriate rate. By the method of "discounting" we can assign different importance to benefits and costs depending on when we receive them. In principle, the farther we experience a gain or a loss, the less important it will be in our present analysis. The discount rate then, reflects the "preferences for present over future consumption" (Freeman III, 2003).

Although there are growing examples of countries and institutions in which CBA is used as a standard tool to assess projects or policies, there are still several European countries in which CBA was never used, while entire sectors are moving towards the use of alternative methods (Hanley, 2000). The main objections raised against the use of CBA for assessing environmental (and some social) impacts derive from the very assumptions in which it is founded.

First, several ethical and moral objections have been put forward against the assignment of a monetary value on the environment. Therefore, the use of strictly technocentric methods like CBA, based on valuation, have been strongly questioned (Mason, 1999). Many people believe that human-centered environmental ethics may not be enough to prevent environmental damage and destruction (Elliot, 2003) and that nature has an intrinsic moral and aesthetic value that can not be measured, let alone traded (Sagoff, 1991, according to Fisher, 2003). Traditional CBA tests exclude as "protest bidders" those who are not willing to accept any compensation at all for a given change in an environmental good because they hold rights-based beliefs. This exclusion is necessary to preserve the theoretical integrity of the method, based on the assumption that everybody should accept eventually some kind of compensation (no matter how high it might be) in exchange of environmental or social losses (or costs). However, excluding people from the analysis is discriminatory and authoritarian (Hanley, 2000). Sagoff (2000) proposed that assigning instrumental value to nature, in order to protect it does not make sense from both the economic and the environmental point of view. In his words, "the question before us is not whether we are going to run out of resources [but] whether economics is the appropriate context for thinking about environmental policy". He advocates that even if we could keep or increase our level of consumption we should not do so for moral, religious, and spiritual reasons. Valuation of environmental goods is sometimes defended by saying that "most valuations… deal with relatively small, non-catastrophic changes in the state of environmental assets, where the advantages of using monetary measures are notable" (Shechter, 2000). This argument ignores or, at least, underestimates the complexity and interdependence of environmental matters, something that goes beyond questions of mere scale. Besides, it completely overrules the possibility of applying the precautionary principle. Several questions remain unanswered: What is "small"? What method is then recommended for "big" issues? When are we going to start applying the alternative method and who will? As said earlier, sustainability is more a pathway than a goal, and requires stepwise changes, no matter how small. If we don't judge our decisions on the basic principles of sustainability (instead of on criteria like economic efficiency) even in the smallest of cases, many of our actions will continue to be part of the problem, and not part of the solution.

Secondly, according to Hanley (2000), CBA copes "rather badly" with the uncertainty associated with complex ecosystems.

Third, although the discount rate is a political decision, the basic assumption behind the method of discounting is that net present value must be maximized, and this "lays potentially heavy costs on future generations" (Hanley, 2000). In fact, at any (reasonable) non-zero discount rate, the present value of damages expected far in the future could be neglected when confronted with present benefits (Freeman III, 2003). On the other hand, the fact that compensating future generations may be impossible challenges one of the very foundations of CBA, which is the possibility that the "winners" can compensate the "losers" and still be better off with the changes produced by the execution of the project. Another issue that may arise related to compensation is that the willingness to accept compensation may differ dramatically according to the economic position (wealth and income) of winners and losers (Freeman III, 2003). According to this line of thought, poor people would tend to accept lower compensations in exchange of natural goods, and this would help perpetuate the present state of inequitable distribution of wealth.

A fourth problem is what Hanley calls "institutional capture". When CBA is used as a standard test within institutions and organisms, the risk of bias increase, making external inspection desirable to check the objectivity of the results. However, this may be practically difficult (if not impossible) in the majority of cases.

Finally, there is the issue of sustainability. CBA is concerned with economic efficiency; sustainability with equity. In the words of Hanley, "…subjecting projects and policies to a CBA test is not a test of their sustainability [because] CBA explicitly allows trade-offs between natural and man-made capital, and thus can lead to violations of the so-called 'strong sustainability' criterion". This criterion requires the maintenance of natural capital, and rejects its replacement by man-made capital. Holland (2003), while recognizing some of the advantages of the economic approach concludes that sustainability "cannot be understood in terms of purely economic criteria".

Economists insist that CBA still can (and have to) play an important role in the decision-making process related to protecting and improving health, safety, and the natural environment, although formal CBA "should not be viewed as either necessary or sufficient for designing sensible public policy" (Arrow *et al.*, 1996). Main advantages of CBA are that it allows the calculation of benefits and costs, and the identification of winners and losers, it raises the subject about the value of environmental goods and services, and it allows the public to express its preferences about environmental policy making (Hanley, 2000). Additional measures (sustainability constraints, shadow projects, etc.) may sometimes be needed to ensure that projects that passed the CBA test are sustainable. However, to minimize the risks and uncertainties (to put it mildly) associated with the enforcement of mitigation measures (especially in developing countries), a reverse approach could be followed. In fact, if projects entailing irreversible changes in the natural capital or unacceptable trade-offs between natural and man-made capital are first rejected on the basis of well-chosen and evaluated sustainability criteria, then CBA can be used for the comparison of the remaining projects. Only in this case, some of the alleged disadvantages of the method would not apply.

EXAMPLE: SEWAGE TREATMENT SYSTEMS

INTRODUCTION

"The present situation (in Latin American countries) is not consistent with sustainable development" (WCED, 1987). Although rich in resources, many Latin American countries (included Argentina) failed to develop for a number of reasons, among which the degree of inequality (especially in the distribution of land and political power) was seen as one of the most important (Arocena and Sutz, 2003). In Argentina, the external debt, a cause and a consequence of failed domestic and international economic and social policies of the past, has also impacted negatively on the process of development. Structural adjustment programs (SAP) devised in international organisms to allegedly rescue the country from economic mismanagement have contributed to deepen the previous crisis and boost administrative corruption. Inequalities within the Argentine society have intensified, the gap between rich and poor has increased, and the "trickle-down" effect is still to show up. According to the WCED (1987), the most important conditions to be met in a country pursuing sustainable development are the following: (a) adequate policies and laws; (b) a proper institutional framework; and (c) the availability, development, or adoption of appropriate technology. In Argentina, leaving aside global economic effects and geopolitical factors, it is the opinion of the author that the weakest points are the lack of coherent and long-term policies (due to political instability and corruption), and the continued use of unsustainable technologies imported from industrialized nations without any adaptation to the local reality. On the other hand, the basic laws and institutions needed to promote sustainable development are already present in the country, although a lot of optimization still needs to be done. Problems from inadequate sanitation and untreated or poorly treated municipal discharges are common in lower and middle-income countries (Elliot, 1999), and Argentina is no exception. These problems generally affect first and harder the less privileged groups of society, increasing and intensifying the misery and life-threatening risks associated with poverty.

Anaerobic wastewater treatment

Organic pollutants in sewage can be biologically degraded via aerobic or anaerobic mechanisms. Anaerobic treatment was used to some extent in the first half of the century but the predominance of aerobic methods became overwhelming later because, among other reasons, an efficient anaerobic system was not available at that time (van Haandel and Lettinga, 1994). Anaerobic processes were "rediscovered" by the introduction of the UASB (upflow anaerobic sludge bed) reactor (Lettinga *et al*., 1980). Although this was a major invention in the field of wastewater treatment, ancestors of this type of reactor can be found in the anaerobic contact process, the Imhoff tank, and the reversed flow Dorr-Oliver Clarigester (Lettinga, 2001). The advantages of anaerobic treatment systems have been thoroughly discussed in chapter 1. The existence of technological paradigms and trajectories in the field of wastewater treatment, social rigidity, and frustrating short-term self-interest in many sectors of society made the introduction of anaerobic technology slower than expected judging by its many potential benefits (Lettinga, 2001). It is clear that there are not merely technological, but particularly also social and economical bottlenecks which prevented the wider dissemination of anaerobic technology (Lettinga, 2001). In some developing countries, negative experiences in the early stages prevented even further the introduction and diffusion of anaerobic systems for sewage treatment. This setback was not overcome until successful full-scale plants were in operation (Conil, P., 2001, personal communication), similar to what was observed in the Netherlands with the introduction of manure digestion (Zeeman, G., 2004, personal communication). Anaerobic technology is already widely applied for sewage treatment in tropical regions and there are clear indications that it could also be used in subtropical countries (see chapter 1). Although some comprehensive "feasibility" studies have been reported, the sustainability of this technology has hardly ever been assessed in depth.

Assessment of the sustainability of sewage treatment systems

It is clear that new research actions are needed to comprehensively understand urban water and wastewater systems from scientific, technical, environmental, social, and economic points of view (Bertrand-Krajewski *et al.*, 2000; Hellström *et al.*, 2000; Balkema *et al.*, 2002). According to Alaerts *et al.* (1990), a wastewater treatment system (at off-site scale) is feasible if it is efficient, effective, reliable, and technically manageable. Based on these general terms, some feasibility criteria were defined, namely: (a) environmental feasibility; (b) reliability; (c) institutional and technical manageability; (d) cost and financial sustainability; and (e) possible application in re-use schemes. Each criterion was further divided to include a wide variety of parameters that need to be considered in the feasibility assessment. Boshier (1993) studied three cases in New Zealand in which communities had to decide on appropriate technology for sewage treatment and disposal. He concluded that the most useful criteria to assess different technological options were: (a) community participation and commitment; (b) availability of disposal sites; (c) physical, cultural, and environmental aspects; (d) environmental risks; (e) costs; and (f) technical aspects. Different technologies were selected depending on the community. In these case studies, cultural aspects played a decisive role in the selection of disposal methods. Dunmade (2002) proposed several indicators to assess the sustainability of a foreign technology (not only sewage treatment technology) for a developing economy, and classified them in primary and secondary. Adaptability of the technology to a new specific location and society was considered the main primary indicator. Secondary indicators were grouped in four major categories: (a) technical; (b) economic; (c) environmental; and (d) socio-political sustainability. By identifying and examining sustainability indicators in a site-specific context, (more) sustainable technologies can be selected and an "enormous waste of economic resources can be avoided" (Dunmade, 2002). Balkema *et al.* (1998) presented a review of the criteria used to compare the sustainability of different wastewater treatment options. The reviewed publications did not use a standardized and comprehensive set of sustainability criteria, and the following new set was proposed: (a) functional criteria; (b) economic criteria; (c) environmental criteria; and (d) social and cultural criteria. These criteria were divided in a number of operational parameters. Lettinga (2001), while criticizing the current states of affaires in industrialized countries with regard to the achievement of a really sustainable world listed some relevant long-term sustainability "commandments" to be met by environmental technologies: (a) little use of resources/energy; (b) production of resources/energy; (c) efficiency and durability; (d) flexibility in terms of scale of application; (e) simplicity in construction, operation, and maintenance. As shown, there are many similarities between criteria proposed by different authors to assess the feasibility and sustainability of sanitation technologies and sewage treatment systems in different places.

MATERIALS AND METHODS

System boundaries and basic assumptions

In this example, the sustainability of different sewage treatment systems was assessed for the city of Salta, in northwestern Argentina (see chapter 2 for more details about the location). The provision of drinking water and sanitation has been privatized and only one company (Aguas de Salta S.A.) serves the whole province. The city's sewerage system covers about 85% of the population. Septic tanks coupled to leaching pits are the main sewage disposal system for the remaining 15%. More than 80% of the collected sewage is treated in an aerobic sewage treatment plant with trickling filters as secondary biological treatment, while the remaining sewage is treated in a system of several waste stabilization ponds (WSP). Both treatment plants are currently overloaded due to the increase in population. A thorough characterization of the city's sewage was presented in chapter 2.

For the sake of this assessment, it was assumed that a sewage treatment plant had to be designed and constructed in the same location of the aerobic plant currently in operation. The plant will have to serve up to half a million people in a projected life span of about 20 years. Domestic sewage will be separately collected in the sewerage, with urban runoff and rainwater conveyed to a different drainage system.

The treated effluent has to be discharged into a nearby river (Río Arenales), which eventually flows into a huge artificial lake (Dique Cabra Corral) 80 km away from Salta city (it is the second most important artificial lake in the country). The water was dammed primarily for the generation of energy and for irrigation purposes downstream, although it is also a growing tourist attraction. Problems of eutrophication have already been observed in this lake, although the ultimate causes of these events are still subject to debate. The river is used for irrigation all the way down to the lake and it is also a source of drinking water for small towns and particularly for people who can not afford a proper connection to the water network.

The assessment has to start when the collected sewage arrives at the treatment plant. The plant would then be inserted in a centralized, off-site, transport-based, sanitation strategy (Alaerts *et al.*, 1990) common to most countries in South America. However, the term "strategy" implies a rational decision-making process that is generally lacking in many of these countries. The sustainability of this very strategy has been recently questioned in Lens *et al.* (2001), with arguments in favor of more decentralized sanitation systems based on the separation at source of different streams and the reuse of by-products. The assessment of the sustainability of the entire sanitation strategy in the city was beyond the scope of this work. However, as it was argued above, even in a centralized context, the selection of treatment technologies should be based on the (comparative) sustainability of the different options. Sewer systems are often constructed and put into operation long before a treatment plant is even planned. The discharge of this collected but untreated sewage into surface and groundwater creates acute environmental degradation and public health threats. In these cases, there is an urgent need to face this immediate problem, while a deeper discussion is held in the sense of deciding what should be the sanitation strategy of the future.

The technologies

The sustainability assessment was confined to the comparison of three different technological options for sewage treatment (**Figure 1**), as follows:

- Option A: *Aerobic high-rate treatment system*. This option consisted of the following units: (a) primary sedimentation tanks; (b) trickling filters (secondary treatment); (c) secondary sedimentation tanks; (d) sludge digestion; (e) chlorination (**Figure 1**, top).
- Option B: *Waste stabilization ponds (WSP)*. A series of three ponds was considered: (a) anaerobic pond; (b) facultative pond; and (c) maturation pond (**Figure 1**, middle).
- Option C: *UASB reactor with posttreatment in polishing ponds*. A single-stage UASB reactor followed by a series of small WSP called "polishing ponds" designed for pathogen removal (**Figure 1**, bottom).

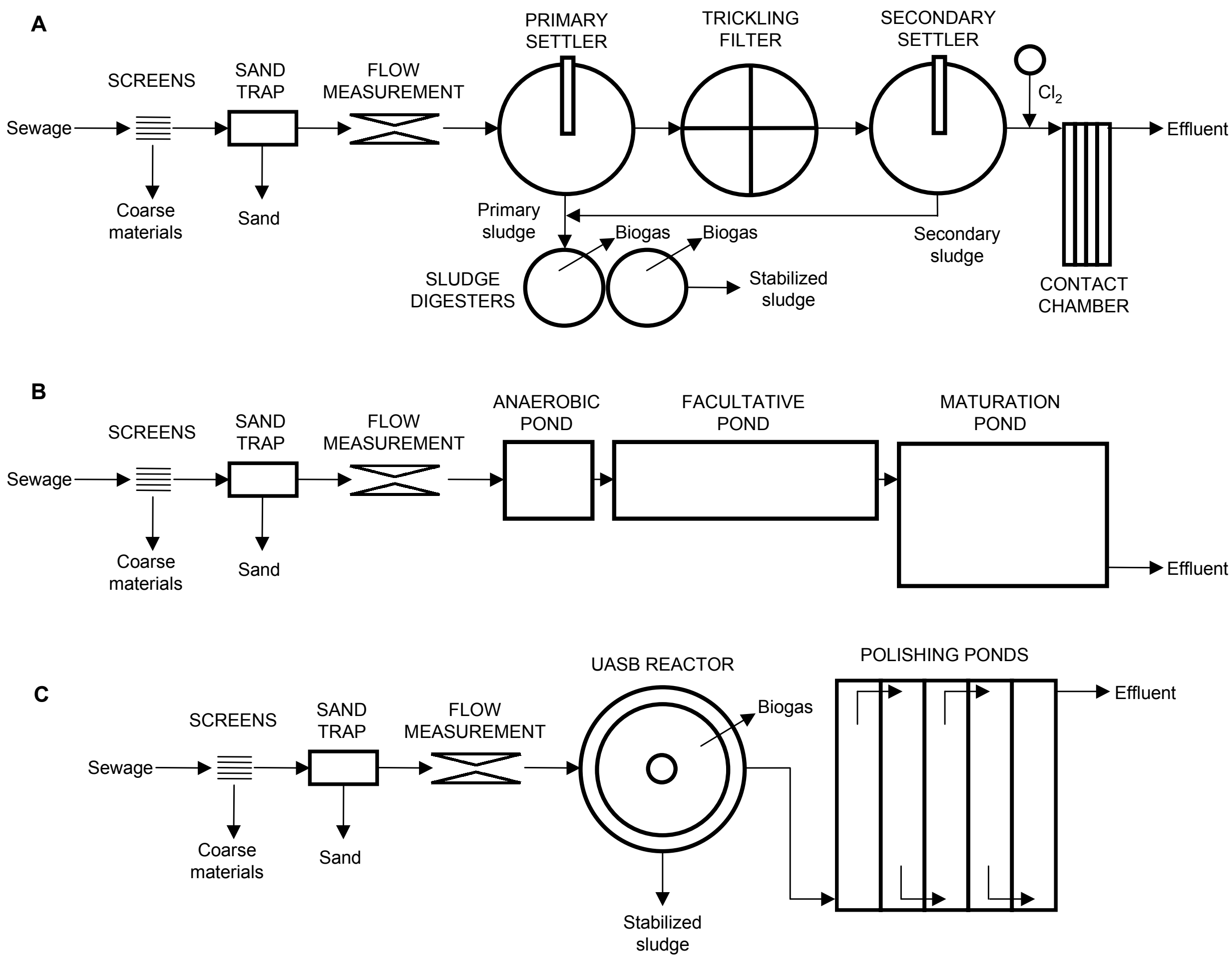

Figure 1. Technological options compared. Not to scale.

The selection of these options was based on their immediate availability in the region. Different assessments can be made with the inclusion of other technological options. However, it was believed that the three options reflected the most realistic alternatives in the local context. Further technical details about these technologies can be found in Metcalf and Eddy (1991), Yáñez Cossío (1993), van Haandel and Lettinga (1994), von Sperling (1996), von Sperling *et al.* (2001), Cavalcanti (2003), and Alexiou and Mara (2003). Besides, von Sperling (1996b) presented a thorough comparison among the most frequently used systems for wastewater treatment in developing countries, including trickling filters, stabilization ponds, and anaerobic systems, among others. Information on the performance of the three options under local conditions was obtained from Liberal *et al.* (1998), the company Aguas de Salta S.A., and results presented and discussed in chapter 5.

The assessment methodology

Sustainability categories

Technologies were compared according to 4 basic criteria subdivided into 20 operational indicators adapted to the local situation from Alaerts *et al.* (1990), Dalal-Clayton (1993), Boshier (1993), Wicklein (1998), Balkema *et al.* (1998; 1998b; 2002), Lindholm and Nordeide (2000), Lettinga *et al.* (2001), Dunmade (2002), and Sanders *et al.* (2003) (**Table 1**). Criteria and indicators intended to satisfy the requirements set by George (1999) to test the sustainability of development projects and the priority actions proposed for the promotion of sustainable sanitation at the second international symposium on ecological sanitation (Lübeck, 2003). From the selection of indicators, it can be seen that even in this relatively small case, a number of global issues, with potentially long-term effects, were identified (for instance, in indicators *Biodiversity* and *Emissions*).

Table 1. Criteria and indicators used to assess the sustainability of sewage treatment technologies.

Criteria	Indicators	Short description[a]
Technical Aspects	Effectiveness	Compliance with discharge standards
	Removal efficiency	Removal of pollutants (when not in standards, or beyond them)
	Reliability	Robustness; vulnerability and risks associated with errors, disasters
	System manageability	Operation and maintenance; reparations; personnel requirements
Environmental Aspects	Conservation	Protection of (fragile) ecosystems and conservation of biodiversity
	External inputs	Need of materials, equipment, electricity, fossil fuels; self-sufficiency
	Land use and impact	Footprint (area occupied); impact on the landscape
	Emissions	Substances released into the environment; pollution prevention
	Reduce, reuse, recycle	Sludge; biogas; treated water for irrigation; nutrients
Social Aspects	Institutions and politics	Basic institutions; awareness in policy-makers/public about sanitation
	Management capacity	Governmental and private proficiency to manage sanitation systems
	Management scale	Operation at different scales and by different actors; decentralization
	Change of routines	Changes by practicioners to adopt sanitation technologies; lobbies
	Social acceptability	Cultural aspects; users adaptation; alleviation of poverty; minorities
	Scientific support	The role of universities and research centers (monitoring, innovation)
	Regulatory framework	Local legislation that promotes or hinders the use of different options
Economic Aspects	Investment costs	Construction costs; equipment required; cost of the land
	Running costs	Operation and maintenance; reparations; availability of spare parts
	Life time	Lifetime of construction items and electromechanical equipment
	Externalities	Changes in natural capital; excavations; social disruptions

[a]Full description provided in the text.

Calculation procedure

Calculations were performed with the methodology used in weighted-scale checklists and results were presented in a matrix. The matrix was based on the "Leopold matrix" first developed in the 70s at the US Geological Survey and the weighted-scale checklists used for environmental impact assessment (EIA) developed by the Battelle Columbus Laboratories in 1973 (Conesa Fernández Vítora, 1997; Modak and Biswas, 1999). The term "matrix" does not have any mathematical implication, but is merely a style of presentation. Weighted-scale checklists recognize the relative differences between the importance of environmental issues, and permit a (quasi) quantitative comparison between technological alternatives according to their potential impact on the environment and society (Modak and Biswas, 1999). The categories were weighted according to their relative contribution to the whole environmental and social system in the local context (Importance). The assignment of a (subjective) importance to each criterion and indicator was done independently of the technologies compared, but according to the social and cultural conditions of the site. After the categories were weighted, a relative score was assigned to each technological option in relation to each indicator (Performance). The individual sustainability for each indicator was calculated as the importance of the indicator multiplied by the performance given to the technology for this particular indicator. Individual scores were summed by row to obtain a "sustainability index" (SI) for each technological option. **Table 2** describes in detail the calculation of the SI. Technologies were finally categorized according to the following ranking: $SI \leq 25$ = very low sustainability; $25 < SI \leq 50$ = low sustainability; $50 < SI \leq 75$ = medium sustainability; and $SI > 75$ = high sustainability. In this way, not only a comparison between the alternatives is possible, but also a sort of "absolute" idea of their sustainability can be estimated, at least for this specific location. The fact that the SI can be understood as a percentage of the sustainability of a (probably non-existent) technology 100% sustainable makes the conclusion of this assessment straightforward and easy to understand.

In this study we assigned importance and performance ourselves, based on personal experience in the region. Although it is obvious that some of the values contain a significant degree of subjectivity, it is also true that many of them have been based on the comprehensive research effort carried out along several years to complete this thesis. Therefore, although some assessments were subjective, they were by no means "blind guesses" but rather "highly educated guesses". However, it is important to stress here that, before real policy decision have to be taken, a similar assessment must be performed by a representative group of local experts and stakeholders in participatory workshops or via consensual techniques like the Delphi method (Linstone and Turoff, 1975; Gordon, 1994). Biases are still unavoidable in this type of assessment due to its subjective character, depending on the persons who perform it. However, subjectivity can be controlled or made explicit and its effects reduced. The selection of participants must be careful to ensure proper representation of all stakeholders, and the information provided to them must be accurate and sufficient to make informed decisions. Communication of the results to all members of society in a simple way is also mandatory to preserve the principle of intra-generational equity (George, 1999). The SI is reliable for comparative studies, because the same group of persons assign importance and performance simultaneously to different alternatives, for the same environmental, social, and economic context. However, because of its local specificity, absolute or general conclusions must not be drawn from the results of such assessments. Comparisons of SIs for the same technology between different locations must be done with care.

Table 2. Detailed procedure for the calculation of the SI. Columns referred to in this table belong to **Table 3**, which was used as an example.

1. The assignment of importance begins with the criteria, and continues with the indicators (column 3). The maximum importance (100) is assigned to the criterion that is judged most important by each participant on the basis of his or her experience and knowledge. After that, relative importance values are assigned to the rest of the criteria. More than one criterion can have the maximum importance. If all the criteria are equally important, all must be assigned the maximum importance. If a criterion is considered irrelevant in the specific context, the importance 0 must be assigned to this criterion, and it could eventually be dropped altogether from the assessment. The assignment can be simplified by reducing the possible relative importance values to only five alternatives: 100, 75, 50, 25, and 0.
2. The criterion with the highest mean relative importance (not necessarily = 100 if values were assigned by more than one person) is then given importance = 100. The rest of the criteria are given proportional importance, calculated as the mean of the criterion divided by the mean of the criterion with the maximum mean relative importance and multiplied by 100. The same procedure is repeated with the indicators (column 4). This step is necessary to subsequently express all categories in percentage units.
3. Importance of the criteria is then transformed to percentage in column 6 by dividing each proportional importance by the sum of the proportional importance for all criteria (subtotals in column 5) and multiplying it by 100, which is the maximum possible importance of the entire system. The importance of each criterion is then a percentage of the total possible importance of the system.
4. This procedure must be repeated for the indicators within each criterion, taking into account that the maximum possible importance attainable by a single criterion is not 100, but the percentage obtained in the former step. After the assignment of importance, each indicator will also be expressed as a percentage of the entire system. The sum of the importance assigned to all the indicators must be 100 (total at the bottom of column 6). Column 7 shows that the sum of the importance within the criteria is equal to the values calculated previously in column 6.
5. When the assignment of importance is completed, the different technologies must be compared. To do so, a certain performance is assigned to each technology in relation with each indicator. The performance reflects the goodness of the technology related to the indicator. The performance can go from 0 to 100, according to the perceived sustainability of each technology for each particular indicator. The higher the performance, the more a particular indicator will contribute to the overall project sustainability. For instance, if a technology is able to provide a very low effluent concentration, the indicator *Effectiveness* for this technology should be assigned a performance = 100. When the performance is 0 for a certain indicator, this indicator must be considered as a “red flag”, or a special case that requires particular attention. Red flags can cause the entire project to fall. For example, if the indicator *Investment costs* has a performance = 0, no matter how high the rest of the indicators might rank, the project is not going to be feasible (let alone sustainable!). The assignment of performance is only done with indicators, not with criteria. The performance of a criterion will be the sum of the performances of the indicators within this criterion. The performance depends on the technology, and must be assigned independently of the importance previously assigned to the indicators. The procedure must be repeated for each technology and by all participants in the assessment (columns 8, 9, and 10).
6. A sustainability score is calculated for each indicator and technology, as importance multiplied by performance divided by the maximum possible performance (100). This is column 8, 9, or 10 for options A, B, and C, respectively, multiplied by column 6, and divided by 100. Results are presented in columns 11, 13, and 15.
7. Columns 12, 14, and 16 show subtotals for each option and criterion, as a percentage of the maximum possible value attainable by the criterion.
8. The sum of all scores for each technology is the “sustainability index” (SI) of this technology. The maximum SI will be 100. SIs are shown as boldface totals in columns 11, 13, and 15.

The SI is a systematic way of comparing alternatives and provides decision-makers with a simple and straightforward tool to back up decisions on the adoption of sanitation technologies in the region. The SI must be calculated and used with responsibility, because sometimes the (excessive) aggregation of data may undermine the transparency of the assessment (Valentin and Spangenberg, 2000). Defining quantitative transform functions for some of the indicators can further deepen the analysis. However, a relatively large amount of good-quality data is generally needed for a quantitative analysis. A qualitative or semi-quantitative approach is deemed useful enough for the purpose of technology assessment and comparison, while keeping its simplicity and readiness to be applied in a broad range of cases.

Why this methodology?

The weighted-scale method was chosen in the example presented in this work, especially over other more structured methods like LCA or CBA, for the following reasons:

a) Simplicity was considered one of the essential features of any sustainability assessment method. The predominant technical nature of LCA and CBA makes it difficult for non-experts to judge whether or not they have been correctly performed (Hanley, 2000).
b) Even when the required technical or economic expertise is available, comprehensive LCA and CBA studies for a project in which there are several environmental and social aspects require extensive fieldwork to gather the necessary information and to assign monetary value to those aspects. The money and time needed to perform this type of research were far beyond the budget assigned to this thesis. Complex and expensive methods could only be applied for large-scale projects for which considerable budgets are allocated. These methods can never be applied for the type of decisions that have to be taken almost on a daily basis, especially in developing countries.
c) The method used has a great potential to include public participation. In fact, a correct implementation of the method must involve an important amount of stakeholders' involvement. Public participation is not required at all to perform either LCA or CBA (beyond the phase of valuation).
d) There are fundamental objections to the use of CBA for sustainability assessment. A CBA test can be performed to compare projects that have passed a previous sustainability test, as discussed earlier. This precaution may override most of the theoretical obstacles that render CBA unacceptable to some people as the sole tool for decision making.

Definition of criteria and indicators

Technical aspects. This criterion refers to the performance of the wastewater treatment system itself.

- *Effectiveness*. Indicates whether the system can comply with local, national or international discharge standards, generally expressed in terms of biological oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), pathogenic microorganisms, and nitrogen compounds, among others, under normal working conditions. Storm water, accidents in the network, natural disasters, sabotages, illegal discharges, and other unpredictable events will fall out of this definition. Daily and seasonal variations in the influent characteristics have to be considered in the design of any treatment system. As pointed out by Alaerts *et al.* (1990), it is important to realize that fluctuations in effluent quality are normal in all types of treatment plants. However, yearly average effluent values conceal the fact that, due to these variations, several days per year the discharge standards might be violated. To minimize these violations, plant designers must use lower mean effluent values in their calculations, which implies that the design removal efficiency needs to be higher and, consequently, so will be the hydraulic retention time (HRT) and/or the plant size. Effluent quality from two or more biological steps in series will tend to be better than from a single-step system because subsequent steps can reduce significantly the effluent variability.

- *Removal efficiency*. The removal efficiency of all relevant pollutants was considered separately from the ability of the system to comply with discharge standards. In fact, two systems can comply with the standards, but one can provide higher removal efficiency for a certain component at the same cost, even if this component was not included in the standards (but may be included in the future). High efficiency guarantees consistent compliance with discharge standards.
- *Reliability*. This indicator refers to the robustness of the system, defined as the capacity to assimilate (ordinary) variations in sewage flow rate and composition, the reaction to shocks, and the time needed to restart the system after breakdowns or maintenance operations. Vulnerability of the system, or potential risks associated with human errors (i.e. spillage of chemicals), equipment failures, natural catastrophes, power outages, vandalism, sabotages, etc., were also included in this indicator.
- *System manageability*. Includes relative complexity of operation and maintenance, and dependency on (complex) infra structural services like power and/or water supply. Availability of spare parts, existence of local know-how, and time between repairs can play a capital role in the feasibility of the technology (Dunmade, 2002). The number and type of personnel (skilled and/or unskilled) required by the system was also included in this indicator.

Environmental aspects. This criterion aims at assessing the environmental impact of the different technologies.

- *Conservation*. Environmental issues like the potential degradation of critical ecosystems and the conservation of biodiversity were grouped within this indicator. The fragility of some ecosystems may require treatment technologies that are especially safe and not prone to fail.
- *External inputs*. This indicator takes into account the need of construction materials, basic and sophisticated equipment, and chemicals (alkalis, chlorine, etc.). The extent of self-sufficiency in construction, operation, and maintenance must be considered here.
- *Land use and impact*. Land used for wastewater treatment is a hidden subsidy and a cost to the community even if the land was given for free to the project (the government could have rented the land) (Alaerts *et al.*, 1990). The land requirements and landscape spoiling were included in this indicator.
- *Emissions*. Wastewater treatment plants produce emissions into the air, the water, and the soil. Anaerobic systems produce methane that must be flared or otherwise used to avoid its release into the atmosphere, where its greenhouse effect is much more important than that of carbon dioxide. However, when methane is used as a fuel, not only its emission is prevented, but also the use of fossil fuels and the concomitant emission of carbon dioxide are avoided. Having said that, it is also important to take into account the presence of dissolved methane in the effluent, especially at low temperatures. If not recovered, this methane will be released into the air when the effluent is discharged. On the other hand, systems that require electricity to operate are indirectly generating emissions at the generation point. Odor nuisance can be a problem of some importance either in aerobic or anaerobic treatment systems (Alaerts *et al.*, 1990). Activated sludge plants can produce aerosols (fine water spray) which can carry pathogens for considerable distances. Noise nuisance can be an issue for some aerobic treatment systems requiring heavy pumping. Emissions to surface water are mainly produced in the form of effluent. The intensity of this emission will very much depend on the treatment system. Even assuming that all systems comply with discharge standards, the higher the removal efficiency, the lower the emission of pollutants and the environmental impact of the discharge. Soil

pollution can arise from inadequate disposal or reuse of treated sewage and biological sludge. Leakage from the bottom of extensive ponds can affect both the soil and the ground water table. Significant acute pollution can result at the point of discharge when a treatment plant needs to be stopped for reparations and maintenance, or due to equipment failure or breakdown. Back up systems, storage ponds, or other mitigation measures (i.e. the setting of an early alert system) may have to be considered when the system is designed. Complex technologies will be more likely to fail and produce acute pollution events, especially in developing countries, where the time between breakdown and reparation can be very long. Within this indicator, it is important to assess the options in terms of their role in the potential prevention of environmental pollution problems.

- *Reduce, Reuse, Recycle*. The production of well-stabilized biological sludge that can be used as a soil amendment or fertilizer would be a positive feature of any wastewater treatment system. However, sludge handling should be safe, simple, and relatively inexpensive compared to the overall running costs. The use of the biogas produced by anaerobic systems can mean significant savings. The treated wastewater is also a by-product, as it can be used for irrigation. In this sense, the removal of nutrients like nitrogen and phosphorus is detrimental. The potential persistence of enteric parasites must be taken into account in reuse schemes, as pointed out by Boncz (2002). Sanitation technologies should aim at the "complete utilization of all possible waste resources" (Lettinga *et al.*, 2001). A proper final destination must be found for all types of residues that can not be reused.

Social aspects. This criterion takes into account social, political, and cultural aspects related to the potential acceptability of the new technology in the local context, and the possibility that it might be definitely incorporated as a current practice by the new users.

- *Institutions and politics*. Basic institutions are needed to promote and manage adequate sanitation systems. Awareness and commitment in individuals working in those institutions are very important factors for the successful implementation and maintenance of sanitation networks. The public in general, and policy-makers in particular, also need to be aware of the fact that appropriate means have to be allocated to plan, construct, maintain, and improve sanitation infrastructure. These considerations are valid for all types of treatment technologies, although new technologies may be more negatively affected (Alaerts *et al.*, 1990). Some technologies are preferred over others based on their local availability, previous successes, and many other (sometimes very subjective) reasons.
- *Management capacity*. There must be a minimum management capacity both at governmental and private level for the successful development of any wastewater treatment technology. Private firms need this capacity to participate in biddings for the construction of sewerage networks and treatment plants, and governments should be able to set adequate technical standards, evaluate bids, and enforce compliance of contract conditions.
- *Management scale*. This indicator refers to the potentiality of the systems to be applied at different scales (off-site, on-site, community on-site), in different areas (urban, peri-urban, rural), and by different actors (governments, private companies, end-users), and the potential of the systems to be applied in a decentralized way. The systems need to be flexible (not scale-specific) in order to adapt to the infinite variations that can be found in real cases. Inflexible systems will tend to force the development of land use and housing according to a pattern that best fit their needs (for example, centralized sewage treatment systems would force the construction of an extensive and expensive collection network even in places where it may be avoided).

- *Change of routines*. Refers to changes needed in the current practices of environmental engineers and experts to adopt (new) sanitation technologies. The more changes needed, the more difficult the adoption. Special attention should be paid to the possible existence of associations or groups of professionals and companies that could resist the introduction of a new technology because they protect preexisting commercial interests related to already established technologies.
- *Social acceptability*. Acceptability of a certain sanitation technology will be a function of society's judgement of its importance (Dunmade, 2002). The importance of certain goods or services can sometimes be associated to the people's willingness to pay for them. The community should financially contribute on a regular basis to a central governmental, a semi-governmental authority, or to a private company for the service of sanitation. In most cases, when this service is centrally provided, paying for it is compulsory. Special attention should be devoted to the presence of cultural aspects that may promote or hinder the spread of a certain technology (attitude towards centralized sewerage versus decentralized on-site sewage management, reluctance to contribute to maintenance operations, sensitivity to odors, willingness to live close to a treatment plant, health aspects, religious principles, current practices and standards of cleanliness and comfort, among others) (Boshier, 1993; van Vliet and Stein, 2003). The existence of active environmental and social non-governmental organizations (NGOs) can be important to facilitate the process of social acceptance by all social actors. Minorities should always be taken into consideration and should participate in the decisions, especially when they could be potentially affected (George, 1999). Public participation of all stakeholders in the decision-making process is essential (including the planning, design, implementation, and monitoring process). The potential contribution to the alleviation of poverty and the improvement of public health, especially for those fractions of the population who are less privileged, must also be an issue for the potential acceptance and adoption of a given technology.
- *Scientific support*. The role of universities and other research institutes may be important in raising the issue of sanitation and assessing possible technological solutions for particular problems and locations. Universities and research centers could also play a role in the continuous assessment (and improvement) of the technology once it is adopted. On the other hand, universities may be a hindrance for the diffusion of new technologies if teachers and staff are not aware of developments in the field of sanitation. Engineers teaching at universities many times work also as consultants or contractors, and may belong to the commercial establishment that opposes new technologies due to sheer ignorance, fear of change, or vested interests.
- *Regulatory framework*. The use of some technologies may require previous adaptation of the local legislation in order to be applied. Others may be already embodied in technical standards and norms. Although the existence of a favorable regulatory framework is not a direct indication of the sustainability of a technology, it may certainly promote or hinder its swift adoption and dissemination.

Economic aspects. This criterion assesses the total costs and benefits of the new technology, taking into account its entire lifecycle and hidden costs that are not included in traditional assessments. These aspects may also be integrated in a cost-benefit framework using CBA as a decision-support technique.

- *Investment costs*. This is a comparative analysis between construction costs of different alternatives for the same site and economic conditions. Centralized sanitation, with conventional sewerage followed by off-site treatment and disposal requires a high initial investment, in principle the highest of all sanitation options (Alaerts *et al.*, 1990).

- *Running costs*. Also a comparative analysis between possible alternatives. Operation and maintenance costs represent an important item in the overall feasibility of the system, and can determine its success or failure altogether. In fact, the lack of operation and maintenance seems to be one of the most widespread causes of technology failure in developing countries (Dunmade, 2002). In many developing countries investment costs are covered by international loans, but operation and maintenance costs, including reparations and spare parts, must be afforded by local authorities. Correct allocation of tax money is then mandatory for a proper operation. A minimum of governmental management capacity and organization is required for that purpose. In this context, systems with low running costs will have more chances of being operated correctly over prolonged periods of time, and may be preferred (Boshier, 1993). Low-cost, locally produced, high-quality spare parts and the immediate and permanent availability of skilled technical experts can be crucial in case of equipment failure or breakdown.
- *Lifetime*. As investment money may not be available again once a wastewater treatment system is built, the longer the lifetime of equipment and construction items, the more attractive a system will become, especially for developing countries. Electro-mechanical equipment and parts are more prone to breakdown and therefore, a sustainable system should avoid them as much as possible. In a traditional CBA test, benefits and costs from different projects are discounted in time to calculate monetary gains or losses occurring at different points in time. However, lifetime is considered to be an indicator on its own because it depends strongly on incidental situations like the availability of international loans.
- *Externalities*. Activities in one part of the social system often generate unwanted (environmental) effects called "externalities" on other parts (Freeman, 1974; Löfgren, 2000). An externality can also be defined as a cost or project output that was not included in the project expenditure and is eventually afforded by the community at large. Externalities can be, in principle, positive or negative, but they are mostly associated to negative environmental effects of economic activities (like pollution). Potential sources of externalities are land excavation, induced ecological change, loss of "natural capital", and any kind of social disruption during the construction and operation of the project (resettlements, destruction of property or cultural heritage, traffic diversion, etc.) (George, 1999; Alaerts *et al.*, 1990). In this study, externalities were related mainly to potential changes in natural capital and disruption of social activities, because more technical environmental aspects (like pollution) were considered under another criterion.

Some indicators could, if necessary, be further divided into more specific factors (i.e. the indicator *Effectiveness* could be divided into factors *BOD*, *COD*, *SS*, *pathogens*, and *nitrogen compounds*, among others). However, the advantage of such subdivision must be justified for each specific case and the costs and difficulties of data gathering must be taken into account. Otherwise, a more general and simple approach seems comprehensive enough to perform an appropriate assessment in most cases.

RESULTS AND DISCUSSION

Table 3 shows the full sustainability matrix, with values assigned to importance and performance and the calculation of the SI for the three technological options. The rationale behind the assignment of values is presented in the next sections.

Table 3. Full sustainability matrix.

1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Assessment parameters		Importance					Performance			Sustainability					
Criteria	Indicators	Mean[a]	Ratio	Total	Value	Check	Option A[b]	Option B[b]	Option C[b]	Option A		Option B		Option C	
Technical aspects		45	47		18										
Environmental aspects		95	100		39										
Social aspects		40	42		16										
Economic aspects		65	68	258	27	100									
Technical aspects	Effectiveness	95	100		6.7		100	100	100	6.7		6.7		6.7	
	Removal efficiency	20	21		1.4		50	75	100	0.7		1.1		1.4	
	Reliability	65	68		4.6		25	100	75	1.1		4.6		3.4	
	System manageability	80	84	274	5.7	18	50	100	75	2.8	62	5.7	98	4.2	86
Environmental aspects	Conservation	35	41		4.8		25	50	75	1.2		2.4		3.6	
	External inputs	85	100		11.8		50	100	75	5.9		11.8		8.8	
	Land use and impact	75	88		10.4		100	25	75	10.4		2.6		7.8	
	Emissions	70	82		9.7		50	25	75	4.8		2.4		7.3	
	Reduce, Reuse, Recycle	15	18	329	2.1	39	50	25	100	1.0	60	0.5	51	2.1	76
Social aspects	Institutions and politics	95	100		3.9		25	100	75	1.0		3.9		2.9	
	Management capacity	40	42		1.6		75	100	25	1.2		1.6		0.4	
	Management scale	50	53		2.0		25	25	100	0.5		0.5		2.0	
	Change of routines	75	79		3.1		75	100	25	2.3		3.1		0.8	
	Social acceptability	50	53		2.0		100	25	100	2.0		0.5		2.0	
	Scientific support	70	74		2.9		25	75	100	0.7		2.1		2.9	
	Regulatory framework	20	21	421	0.8	16	100	100	100	0.8	53	0.8	77	0.8	73
Economic aspects	Investments costs	45	45		4.7		25	100	75	1.2		4.7		3.5	
	Running costs	100	100		10.4		25	100	75	2.6		10.4		7.8	
	Life time	75	75		7.8		50	100	75	3.9		7.8		5.9	
	Externalities	35	35	255	3.6	27	100	25	100	3.6	43	0.9	90	3.6	78
Total					100					**55**		**74**		**78**	

[a]Average of importance values if assigned by more than one person.
[b]Average of performance values if assigned by more than one person.

Assignment of importance

Criteria

Figure 2 shows proportional importance values assigned to the different criteria (from **Table 3**). The criterion *Environmental aspects* was considered the most important of the four criteria for this particular situation. Spread of water-borne diseases, deterioration of water supplies, and environmental degradation (pollution and loss of aquatic biodiversity) were the main reasons behind the perceived need for adequate sanitation. The discharge of untreated sewage to the environment was considered to be one of the most pressing problems of the city and the region at large.

Economic aspects was considered the second most important criterion because governmental budgets are always short for constructing and maintaining basic infrastructure, especially in developing countries.

Technical aspects ranked third. It was believed that the region has or may have a relatively easy access to good-quality technologies, once political and economic decisions are made. On the other hand, technological capability in the region was considered sufficient to deal with almost any kind of sewage treatment technology, irrespective of its technical characteristics.

Social aspects was considered the least critical criterion in the context of the region. Increasing awareness among scientists, policy-makers, private companies, and the society at large about environmental problems has fostered a growing demand for sanitation technologies. In this context, and for this particular case, any type of sanitation technology would be most welcome by society, and will certainly make an important contribution to the alleviation of poverty-related problems. The adoption of a particular technology among the three options compared would not imply a major change in the daily life of the inhabitants, assuming that the sewerage is already in place.

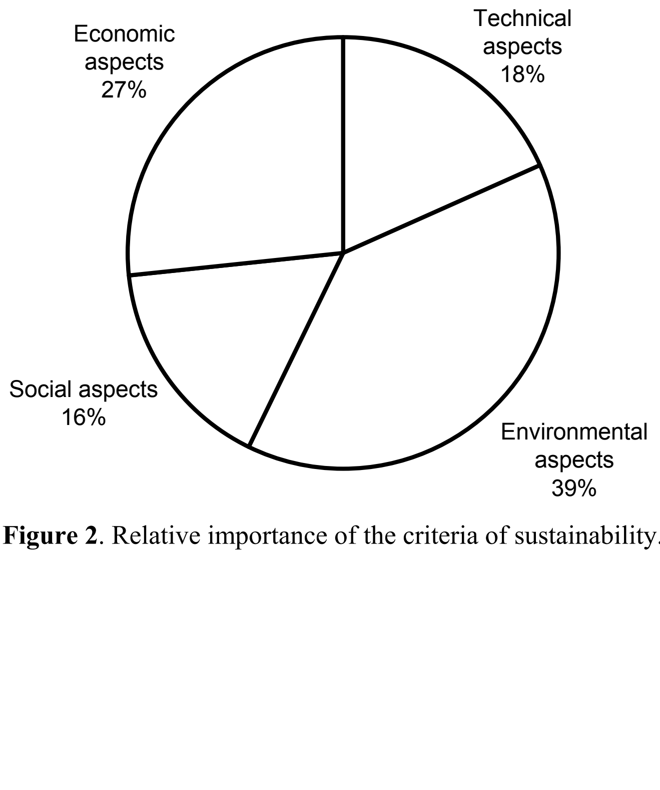

Figure 2. Relative importance of the criteria of sustainability.

Indicators

The most important indicators were *External inputs*, *Land use and impact*, *Emissions*, and *Running costs* (**Figure 3**, based on **Table 3**).

Within *Technical aspects*, it was considered that all possible solutions must provide adequate *Effectiveness* (see column 4 in **Table 3**). This means that compliance with discharge standards (on average) should be guaranteed by all technologies in order to be even considered as a possible alternative. *System manageability* was also considered a priority in the regional context, especially because the need of spare parts and lack of trained personnel, were seen as a possible reason why a (complex) technology can fail to achieve its full potential. *Reliability* ranked third but was also considered relatively important in the context of the region, where common practical problems like power outages may affect negatively the performance of the systems. *Removal efficiency* was only seen as a way of rewarding technologies that are able to provide additional (more than legally required) removal of some pollutants.

Within *Environmental aspects*, it was considered that *External inputs* was the most important indicator (see column 4 in **Table 3**). In developing countries, the need for energy and resources is many times inversely related to the long-term use of a technology. Budget shortages and bureaucracy can delay the supply of raw materials, jeopardizing the operation and usefulness of technologies dependent on these materials. *Land use and impact* was also considered very important around the city, where land is very expensive, contrary to what is generally assumed in developing countries. *Emissions* from the use of any sanitation technology were also important in this assessment. *Conservation* did not rank very high primarily due to the fact that treatment technologies will generally tend to decrease, rather than increase the environmental impact on water bodies. This is particularly the case when the discharge of raw or poorly treated sewage has been a common practice in the past. *Reduce, Reuse, Recycle* was the least important indicator within this criterion. The possibility of recovering useful materials for further utilization is certainly beneficial, but it was not considered a crucial indicator of the sustainability of sanitation technologies in the local context. In fact, agriculture in the region is rarely performed (very) close to the cities, where treatment plants are generally located. Therefore, in the context of the present sanitation strategy, reuse of by-products or treated water would demand additional investments in order to make these products available to relatively distant farms.

Within *Social aspects*, the most important indicator was the *Institutions and politics* (see column 4 in **Table 3**). When policy makers and the public in general are convinced, the process of technology transfer is greatly enhanced and facilitated. The indicator *Change of routines* was considered very important for the successful adoption of any sanitation technology. The presence of adequate *Scientific support* was also considered an important indicator of the sustainability of a technology in the region. In fact, the absence of scientific support would certainly mean that the technology would only be used while external support is provided. In this study, where the size of the treatment plant is relatively fixed as a precondition, the indicator *Management scale* is not very relevant. However, it was given a medium importance mainly because the potential of a treatment technology to be applied at different scales was considered very important in terms of the sustainability of the technology beyond the time span of a single project. *Social acceptability* was considered relatively guaranteed for any kind of sanitation technology. No major cultural issues were identified that could oppose the introduction of sanitation technologies, especially because alleviation of poverty-related problems (like health risks posed by water-borne diseases) would be associated to any sanitation technology adopted. Similarly, a reasonably good level of *Management capacity* is present (or easily achievable if the political will does exist) both at governmental and private levels, ensuring that any introduced technology will be adequately handled. The *Regulatory framework* is adequate and general. There are no particular sanitation technologies embodied in the norms.

Running costs was considered to be the most important indicator of sustainability within *Economic aspects* because this item will impact directly on local people through taxes or the payment of public services. *Lifetime* was also considered a very important indicator. The longer a system can work, the more sustainable it will be, even if the technology becomes somewhat obsolete. International loans are only granted after long and painstaking procedures, and it is unlikely that loans will be granted twice for the same purpose, i.e. the construction of a sewage treatment plant. *Investment costs* was not assigned a higher importance because the availability of funds will depend greatly on the political decision. However, when the construction funds need to come from the government (and not from an external loan), this indicator may become the most important indicator within the economic aspects. *Externalities* was considered to be the least important indicator within this criterion, especially because the assessment was comparative. In particular cases, however, this criterion can be extremely important, namely when the use of a particular technology can destroy, or in any way irreversibly alter ecosystems, biodiversity, cultural heritage, and so on.

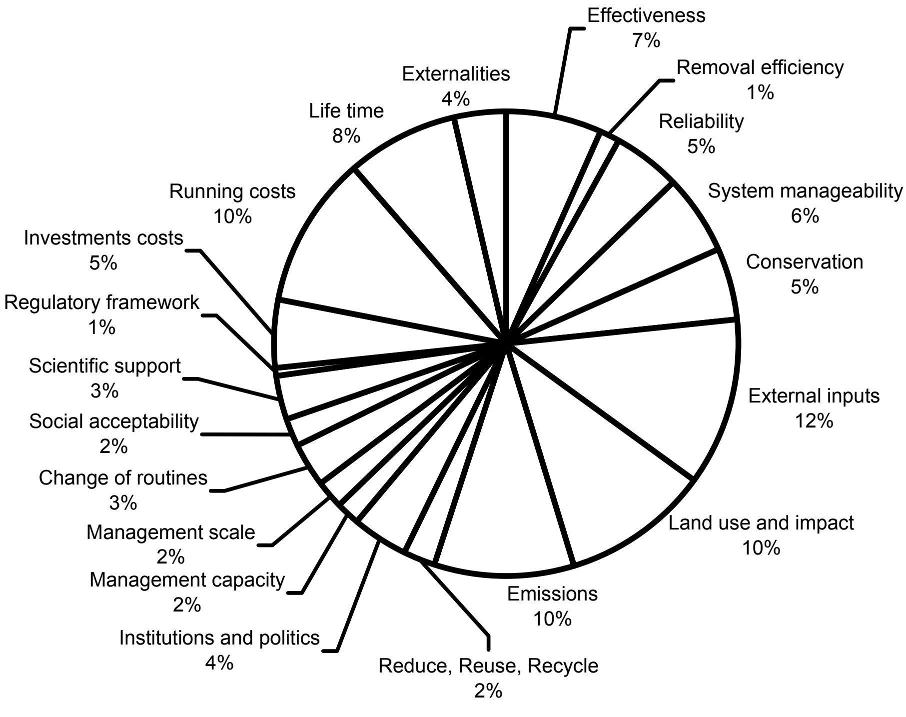

Figure 3. Relative importance of the indicators of sustainability.

Assignment of performance

The assignment of comparative performance for the technologies assessed was done step by step for all the indicators. Results are shown in **Table 3** (columns 8, 9, and 10). The main reasons behind the assignment of performance are presented in the following paragraphs.

- *Effectiveness*. All technologies were assumed to be able to comply (on average) with discharge standards.
- *Removal efficiency*. Reports on the efficiency of different treatment systems are very diverse (von Sperling, 1996b), making comparisons difficult. Local results suggest that option C is the most efficient of the three options, followed by option B, and option A.
- *Reliability*. According to von Sperling (1996) options A and B are resistant to variations in influent characteristics (flow rate and composition). However, as pointed out by Alaerts *et al.* (1990), trickling filters can be sensitive to low winter temperatures in moderate climates. According to their magnitude, temperature and flow rate variations could also affect option A and the anaerobic step of option C. Posttreatment in option C would absorb most of the fluctuations that could occur in the UASB reactor. On the other hand, the higher need of electromechanical equipment (pumps) and chemicals (chlorine) makes option A highly prone to failure. Power outages would have little effect on anaerobic technology, while they are almost completely irrelevant for option B. All in all, option B was considered the most reliable option, followed by option C and option A, in that order.
- *System manageability*. Manageability was considered to be highest for option B because of its simple operation, no need of spare parts or reparations, and relatively little requirement of personnel (either trained or untrained). Attention must be given, however, to the generally neglected maintenance of banks and shores (erosion prevention, grass mowing) and possible need of permanent surveillance of the whole area to avoid vandalism or potential health risks to nearby population. The lower manageability assigned to option C was mainly based on the fact that there is little experience with this kind of systems in the region (although there is a lot of experience in tropical countries). Skills needed to operate option A could be difficult to get in some small towns and cities, making this technology not feasible in those situations. Difficult handling, treatment, and disposal of the high amounts of sludge produced are probably the most relevant aspects in the manageability of aerobic systems like option A.
- *Conservation*. Lower reliability of option A could imply adverse effects on local ecosystems. Option B ranked better, but the discharge with a high content of green algae was perceived as potentially negative for the receiving river which, in turn, discharges into a lake with recent events of algal blooms. Option C ranked well because the quality of the effluent was considered the best (very low concentration of organic matter and algae), although it was regarded as less robust than option B, especially due to lack of local expertise.
- *External inputs*. Option A requires extensive civil construction work, the purchase of expensive electromechanical equipment, and the use of chemicals to disinfect the final effluent. Construction materials are also needed (although a lesser amount) for the anaerobic step of option C. Construction materials are almost not needed for option B. However, artificial liner is increasingly being required to render impermeable the base of WSP to avoid groundwater pollution. Besides, the construction of a fence around sewage treatment plants is generally mandatory for safety reasons and extensive systems will then require a higher investment. Chemicals are not needed to operate neither option B nor C. As long as flow rate management can be done by gravity, options B and C do not require any (significant) use of electricity. Option A is very much dependent of electricity, especially for sludge management.
- *Land use and impact*. Option B was the worst option for this indicator. Extensive areas are needed to build WSP. Option C ranked second, mainly because of the need of posttreatment. Option A was the option with less land requirements. A system anaerobic pond + facultative pond would require 1.5 – 3.5 m^2/inh, compared to 0.5 – 0.7 m^2/inh for a low-rate trickling filter, and 0.05 – 0.10 m^2/inh for a UASB reactor (without posttreatment) (von Sperling, 1996).

- *Emissions*. Within this indicator, it was assumed that all options have the same effluent quality. Option B was the worst option for this indicator, because of unavoidable methane emissions and bad odors from anaerobic ponds, and the risk of leakage to the ground. Sometimes leakage was found to be so severe that some WSP can fail to produce any effluent (Grau, 1996). The need of electricity generates indirect emissions in option A. Chlorination is a controversial disinfection process that can produce undesired secondary compounds, although in many cases it is an effective way of eliminating pathogenic microorganisms (Lomborg, 2001). The production of significant amounts of biological sludge can become a big problem in itself because, if it is not reused in agriculture, it must be adequately disposed of in sanitary landfills often nonexistent in developing countries. Although it was not the case for this particular work, the need of adequate disposal sites involves additional administrative and financial burdens to the plant manager. Biogas production that can be potentially used to reduce the consumption of fossil fuels is a positive feature of option C. However, dissolved methane in the effluent of anaerobic reactors might be emitted to the atmosphere when the effluent is discharged if proper measures are not put in place (for this reason, option C was not given a higher score for this indicator). Biogas is also produced during the stabilization of primary and aerobic sludge in option A. It can be used to (partially) heat the digesters. This biogas may not represent a net benefit, because most of it comes from the degradation of newly formed (aerobic) biomass. Noise nuisance was not expected to be a problem for the technologies compared, if properly managed.
- *Reduce, Reuse, Recycle*. The production of stabilized sludge, biogas, and an effluent containing most of the nutrients made option C the most attractive option under this indicator, followed by options A and B.
- *Institutions and politics*. Environmental agencies have been created in the Province relatively recently (in 1997 at municipal level and in 2000 at provincial level) and there is a renewed interest in sanitation topics. For historical and economic reasons, WSP are generally the preferred option. However, research carried out at the local university, partially funded by a provincial governmental agency (this thesis!) has raised the interest on anaerobic systems. Conventional aerobic systems (option A) tend to be dismissed by policy makers and private companies alike, based on construction and operation costs and manageability considerations.
- *Management capacity*. Local management capacity is considered enough for the adequate development of the three options. Environmental agencies and other governmental control organisms have the professional capacity to carry out biddings and to supervise construction, operation, and maintenance of wastewater treatment systems. There is reasonable local expertise in the design, construction, and operation of WSP and, to a lesser extent, trickling filters. There is interest on the development and application of anaerobic reactors, although there is still lack of trained professionals and technicians on these systems. Low value assigned to option C was based on the lack of local experience.
- *Management scale*. It was considered that economic limitations or environmental risks prevent options A and B, respectively, from being applied at small scale. Option C was considered the only one that could be safely applied at any scale at reasonable costs.
- *Change of routines*. Due to its widespread use in the region, there are no expected (major) changes needed among engineers and practitioners for the use of option B. Option A may need some additional training of personnel before its full-scale application. The application of option C was regarded as the most difficult due to the lack of local expertise.
- *Social acceptability*. Acceptability was considered high for option A and C. Option B was assigned a lower performance based on some cases of public rejection due to odor problems. Acceptability is associated to the willingness to pay for a certain service. As the service of water and sanitation is private, people are already used to pay for this service.

- *Scientific support*. Paradoxically, there is more scientific support for anaerobic technologies than for all other options. Local research has also been performed on WSP, but not on trickling filters.
- *Regulatory framework*. All technologies are equally accepted in the existing norms.
- *Investment costs*. Although land is not cheap at the selected site, option B was still considered the cheapest option, in agreement with von Sperling (1996). On the other hand, anaerobic treatment with posttreatment (option C) was considered to be significantly cheaper than fully aerobic alternatives (option A) (based on Alaerts *et al.*, 1990).
- *Running costs*. Running costs will strongly depend on local conditions (labor and energy costs), as well as on the cost and availability of spare parts and chemicals. Aerobic systems (option A) have in general higher operation and maintenance costs than anaerobic alternatives (McKinney, 1983; Vochten *et al.*, 1988; Eckenfelder *et al.*, 1988; van Haandel and Lettinga, 1994). WSP were considered the least expensive, although it is sometimes due to inadequate maintenance.
- *Life time*. Wastewater treatment systems are usually designed for a lifetime of about 20 years (Metcalf and Eddy, 1991). The presence of mechanical equipment will tend to shorten the lifetime, especially when spare parts may be difficult to obtain. Adequate construction materials are needed for anaerobic treatment to avoid corrosion.
- *Externalities*. Extensive excavation needed for the construction of WSP, the potential presence of water-borne vectors, and a reduction in property value were considered the main reasons for the low score assigned to option B. Options A and C were considered to have a similar level of potential externalities.

Sustainability

Overall sustainability was *medium* (to low) for option A, *medium* (to high) for option B, and *high* for option C (**Figure 4**; see the boldface totals at the bottom of columns 11, 13, and 15 in **Table 3**).

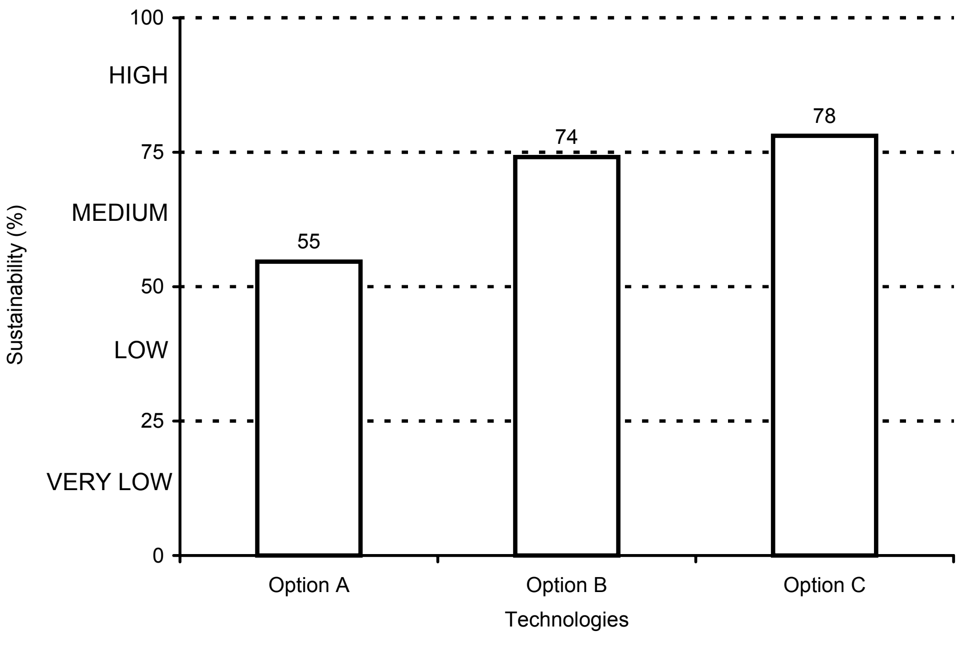

Figure 4. Sustainability index of the different options.

All in all, the difference between options B and C was rather marginal. The fact that option A scored low for *Economic aspects*, especially *Running costs*, with a high importance value (**Table 3**, column 12) can be seen as the main reason why this option obtained the lowest SI. Option A also scored low for *Environmental aspects* (especially *External inputs*) and *Technical aspects*, due to its higher complexity, compared to the other options. This complexity influenced social indicators as well, especially *Institutions and politics*. The main strength of option B lied in its simplicity and low cost (very high values for *Technical aspects* and *Economic aspects*) (**Table 3**, column 14). It did not do well in *Environmental aspects* due to its high demand of land, and the possibility of having uncontrolled *Emissions* into the air and the soil (this could be a major obstacle for its long-term sustainability). Although option B ranked relatively high in *Social aspects* as a whole, the fact that its acceptability was considered low can also be a major inconvenience for the further dissemination of this technology. Option C obtained relatively high scores for all criteria, ranking especially better than the other options for *Environmental aspects* (**Table 3**, column 16). Its main weakness at this point in time seems to be the lack of local expertise, something that have negative influence on several indicators. Based on the results in **Table 3**, the main strengths and weaknesses of the options compared can be easily identified (**Table 4**). A simple list of strengths and weaknesses, together with the SI, can be a very useful way of communicating the results. Strengths and weaknesses derived from the sustainability matrix are a confirmation of the advantages and disadvantages generally reported for these types of treatment systems (van Haandel and Lettinga, 1994; von Sperling, 1996).

Table 4. Main strengths and weaknesses of the options based on results from the sustainability matrix.

Option	Strengths	Weaknesses
A	▪ The system is efficient ▪ Land requirements are very low ▪ Social acceptability is high	▪ High investment and running costs ▪ Operation is relatively complex ▪ Generates a lot of biological sludge
B	▪ The system is very efficient and reliable ▪ Construction is relatively cheap ▪ Operation is cheap and simple	▪ Requires a lot of land ▪ The risk of emissions to the air and the soil is higher ▪ People may reject it due to the potential production of bad odors and vectors
C	▪ The system is very efficient ▪ It is environmentally sound ▪ Generates useful by-products and energy	▪ There is little experience with anaerobic reactors in the region ▪ Investment costs can be high ▪ Companies may resist the adoption of this technology

CONCLUSIONS

- A UASB reactor followed by adequate posttreatment, e.g. polishing ponds, was found to be the most sustainable alternative for sewage treatment in the region.

- The method used was simple to perform and sensitive enough to detect differences in sustainability between the technological options compared.

- Validation by local actors is required before final conclusions are drawn and policy decisions are taken.

FINAL DISCUSSION

The direct treatment of raw domestic sewage with anaerobic technology would be a radical innovation in the context of the region, although anaerobic technology as studied in this paper would not imply a transformation of the entire technological system. However, the high SI obtained for this option is a strong indication that its adoption can entail several benefits.

The validity of the assessment method used in the example is based on its comparative nature, and refers solely to the technologies compared, embodied in a centralized, off-site, transport-based, sanitation strategy. It could be argued that decentralized sanitation systems (in which anaerobic technology can play a central role), with a stronger component of end-users involvement, less requirements of resources and energy, and local reuse of by-products might rank higher in any sustainability scale.

One of the main strengths of the method used is that it can be applied almost irrespective of the amount of information available, although more information would imply a better quality of the results. Further technical, environmental, economic, and financial studies to be carried out by specialists might be necessary to elucidate specific aspects and to adequately inform stakeholders and policy makers about the relative advantages and disadvantages of different options. Economic and financial techniques that may be extremely useful for this purpose are basically related to CBA, valuation studies, and the proper discounting over time of costs and benefits, as discussed earlier. The calculation of indicators like the internal economic rate of return (IERR), the net present value (NPV), or the benefit-cost ratio (BCR) will then be required. Other techniques proposed in literature for the economic assessment of projects are the risk-cost-benefit analysis (RCBA) (Shrader-Frechette, 1985), the calculation of opportunity costs, shadow pricing, and total annual costs per household (TACH), among others (Alaerts *et al.*, 1990). The simplicity and democratic character of the assessment must not be overshadowed by complex and lengthy technical studies that may be viewed as techno-centric black boxes (Hanely, 2000). However, the role of science will always be crucial to provide rational assessment of the uncertainties and risks associated with different decisions (Slingerland *et al.*, 2003).

There is no ideal approach to a complex challenge as the selection of a more sustainable technology for a given situation and location. The method used in this study is just an example of how the various aspects of the sustainability notion can be rationalized through a set of concrete indicators. Still, it goes beyond most of the studies found in literature about this topic in that it actually attempts to provide concrete figures about a real situation. Conclusions from the assessment could be easily communicated to the public and policy makers in the region, but can not be extrapolated to regions with different environmental or cultural conditions. The assumptions made during the assessment must be clear because different assumptions can lead to different conclusions.

No matter how sustainable a technology might seem in theory, the implementation and monitoring phases will be crucial for its success. Strict control of contractors during construction, operation, maintenance, and other procedures is mandatory. A comprehensive long-time monitoring of the system(s) will help to draw definite conclusions for future projects.

The fundamental principles behind the development and use of sustainability assessment methods can be summarized as follows:

1. Sustainability is a criterion that should be considered in decision-making.
2. Sustainability has technical, environmental, social-cultural, economic, and institutional dimensions.
3. Methods to assess the sustainability of policies, systems, concepts, or technologies must be rational, although they can (and most of the times probably have to) be qualitative and subjective and not only quantitative and objective. There is some controversy about the right methods to apply and a wide variety of them have been proposed.
4. The role of science is vital to provide information for an adequate decision-making.
5. Participation of stakeholders, local actors, and (local) experts is required to make a representative assessment.

Policy decisions are taken every day. In the absence of a rational and democratic method, these decisions will continue to be taken in an irrational (or, at best, technocratic) and authoritarian way. The development and use of methodologies to perform sustainability assessments would be a step forward in the direction of more sustainable (urban) water and wastewater systems and a better quality of life for the people in developing countries.

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CHAPTER 7

General summary – Algemene samenvatting – Resumen general

GENERAL SUMMARY

The framework

In the province of Salta, in northwestern Argentina, as in many developing countries, one of the most pressing environmental problems is the widespread discharge of raw or poorly treated sewage into rivers, lakes, open canals, and other watercourses. This practice leads to serious threats to public health, acute environmental problems, and deterioration of the landscape. Economic consequences are, among others, a reduction in property value, the loss of traditional recreational activities like fishing or camping, and the virtual disappearance of revenues from tourism. As usual, the poor and less privileged sectors of civil society are hit first, harder, and some times irreversibly. Actions to tackle this problem have been generally oriented to the construction of big, expensive, and complex (aerobic) treatment systems in some cities, and badly designed and maintained waste stabilization ponds (WSP) in the outskirts of small cities and towns. With increasing urbanization, cheap land is no longer available close to the cities, and WSP need to be constructed at considerable distances, demanding longer and more expensive sewer systems. Besides, some existing WSP are coming under considerable pressure to be relocated farther away due to complaints about odor nuisance and potential transmission of water-borne diseases to the ever closing population.

For the reasons explained above, developing countries are in urgent need for simple, affordable, compact, flexible, and efficient sewage treatment systems. Anaerobic technology seems to comply rather well with these conditions. However, up until recently, it was believed that sewage could only be treated anaerobically in tropical countries, where sewage temperature is generally above 25°C. The climate in Argentina is mainly temperate, and therefore there was allegedly no room for anaerobic sewage treatment in this country. Besides, anaerobic treatment was only considered advantageous when chemical oxygen demand (COD) concentration in sewage was high (say higher than 500 mg/L). Therefore, the feasibility of anaerobic treatment was highly questioned for the rather diluted type of sewage (less than 250 mgCOD/L) that seems to be more the norm than the exception in Argentina.

The first and main technical obstacle for the anaerobic treatment of sewage in subtropical regions using upflow anaerobic sludge bed (UASB) reactors was allegedly the presence of suspended solids (SS). Hydrolysis, the rate-limiting step of the process of anaerobic digestion of particulate organic matter, may become too slow at low temperatures, leading to accumulation of undegraded SS in the reactor's sludge bed. Under these circumstances, the solids retention time (SRT) in the reactor may become too short to provide good treatment efficiency and satisfactory sludge stabilization at a reasonable hydraulic retention time (HRT). Pre-removal of SS may be needed when sewage is treated in UASB reactors at short HRTs and low temperature. The application of two-stage anaerobic systems was also proposed to overcome the problem of solids accumulation under these conditions.

The work

In chapter 1, a comprehensive literature review on the use of upflow reactors for sewage treatment was presented. Laboratory, pilot-scale, and full-scale experiments and applications from all over the world were described and discussed. Some research needs were identified.

A thorough physical, chemical, and biological characterization of different types of sewage from the city of Salta was presented in chapter 2. Parameters measured included temperature, basic composition (pH, organic matter, alkalinity, volatile fatty acids, total and suspended solids, nutrients, etc.), anaerobic biodegradability, and the hydrolysis rate constant of particulate organic matter. A complete description of sewage was considered indispensable as the first step in the assessment of the feasibility of anaerobic treatment in the region.

Results presented in chapter 3 and 4 show the performance of a sewage treatment system consisting of a primary sedimentation tank (settler) followed by a pilot-scale UASB reactor. This was the first experiment ever conducted in the country on anaerobic sewage treatment. The system was operated for more than seven years in the city of Salta, where mean ambient temperature is 16.5ºC, and freezing temperatures are registered during several weeks per year (**Figure 1**). During the experiments, mean annual sewage temperature was 23.0ºC. The minimum registered daily sewage temperature in this period was 12.6ºC. Very high COD removal efficiencies (up to 84% in total COD and 92% in suspended COD) have been observed in the entire system at a mean HRT of 2 h in the settler and 5.6 h in the reactor, equivalent to an upflow velocity (V_{up}) of 0.71 m/h. Big granules developed in the reactor probably due to the low concentration of SS and COD in the influent and an adequate combination of HRT and V_{up} (**Figure 2**). The specific methanogenic activity (SMA) of the sludge was rather low (0.10 gCOD-CH_4/gVSS.d)[1] and probably most of the granules (especially the bigger ones) contained considerable amounts of dead anaerobic bacteria and inert organic matter. The sludge growth rate was very low and it was calculated that only one discharge every two years could be enough to dispose of the excess sludge. The system studied was highly robust and efficient, and consistently delivered a final effluent in compliance with discharge standards for COD and SS. A safe and efficient operation could be achieved in 4-m tall UASB reactors when the sludge bed is kept between 1 and 3 m, and the V_{up} applied is around 0.5 m/h (HRT ≈ 8 h).

Having demonstrated the (technical) feasibility of anaerobic treatment of settled sewage under local conditions, the direct anaerobic treatment of raw sewage was attempted. For that purpose, a two-stage UASB system was built and operated (**Figure 3**). Chapter 5 shows that a two-stage anaerobic system is also highly efficient and robust. A total COD removal efficiency of 89% was obtained in the two-step anaerobic system at HRTs of 6.4 + 5.6 h (V_{up} = 0.62 + 0.70 m/h), with 83 and 36% removal in the first and second steps, respectively. The effluent concentration was similar to that

[1] Methane production, expressed in grams of COD, per gram of volatile SS and per day.

obtained in the system settler + UASB reactor described in chapter 3. Moreover, as the first reactor removed most of the incoming COD at a very reasonable HRT, the second step was virtually redundant. This represented a major breakthrough and meant that direct anaerobic sewage treatment was possible (and, in fact, efficient) under subtropical conditions in single-stage UASB reactors, very much like in tropical regions. The performance of the system was not affected during the coldest period of the year, which usually lasts about three months. The sludge retained in the first reactor showed good stability, especially in summer time, and could be directly disposed of without further treatment. Another interesting observation was that the startup of the first UASB reactor was achieved in only one month, without the addition of any inoculum, at an HRT of 8.2 h, and a V_{up} of 0.48 m/h. SMA in the first and second UASB reactors was 0.12 and 0.04 gCOD-CH_4/gVSS.d, respectively. The sludge growth rate in the first UASB reactor was less than 5% of the reactor volume per month. It was calculated that excess sludge could be disposed of through only one or two discharges per year, minimizing reactor disturbances and potential human errors.

Anaerobic treatment alone does not provide an effluent that can be discharged into a river or water body mainly because the concentration of pathogenic microorganisms is too high. Therefore, a posttreatment step is necessary when UASB reactors are used for sewage treatment. WSP are a cheap and efficient option for this purpose. A system of five WSP in series was studied as a posttreatment system for the anaerobic steps (**Figure 3**). The ponds were only needed for the removal of pathogenic microorganisms and therefore, the design surface area was small, especially because a hydraulic plug-flow regime could be approached. As shown in chapter 5, the concentration of pathogens in the final effluent of the ponds complied with the most stringent discharge standards. The increase in COD concentration in the ponds (due to the growth of algae) did not make the effluent exceed discharge standards for COD. As a general conclusion, it could be said that a single-stage UASB reactor followed by a series of WSPs could be a very efficient, reliable, and simple system for the treatment of raw sewage in subtropical regions like Salta.

Results presented and discussed in chapters 2 to 5 showed that the systems studied were efficient and reliable, and that their full-scale use was highly feasible in the region. However, a purely scientific assessment seemed insufficient to guarantee the acceptance and implementation of this technology on the ground. In chapter 6, it is argued that the notion of "sustainability" can be a comprehensive criterion to assess different (environmental) technologies. The use of this criterion as a decision-making tool has theoretical and moral roots in the concept of "sustainable development" introduced in the political agenda in 1987 by the World Commission on Environment and Development (WCED). A comprehensive sustainability assessment should take into account the global, long-term context in which technology is used because it is argued that there are none or few technologies that can be considered intrinsically sustainable. As a practical example, a comparative assessment of three technological options for sewage treatment was performed for the city of Salta, in terms of a series of technical, environmental, social, and economic criteria and indicators. The following options were compared: (A) aerobic high-rate treatment system (primary and secondary settling, trickling filters, sludge digestion, and chlorination); (B) three conventional WSP in series; and (C) UASB reactors with posttreatment in small "polishing" WSP. The selection of these technologies was based on their immediate availability in the region. In this preliminary assessment it was found that, under local conditions, a system UASB + WSP was more sustainable than the other two options (results in chapter 6). The assessment method used was simple to perform and sensitive to detect differences in sustainability between the options compared. However, this method should not be seen as a blueprint for sustainability assessment. The selection of the most appropriate method for a given case will depend on the type of assessment to be performed and many other site-specific conditions. In chapter 6, the need for rational, simple, and comprehensive methods to assess the sustainability of different technologies is emphasized and discussed. It was also stressed that before a concrete policy decision is actually taken, a representative panel of local experts, actors, and stakeholders must perform the actual assessment in a transparent and participatory way. The final plead of chapter 6, and this thesis, is that sustainable development will only be achieved through a fully democratic way of technology assessment and decision-making that can go beyond political and economic motivations and that may be able to address and solve environmental problems and social injustices.

The people

This thesis could not have been completed without the coordinated effort carried out by the members of the Laboratory of Environmental Studies (Laboratorio de Estudios Ambientales, LEA) (**Figure 4**). Dr. Grietje Zeeman (in the Netherlands) (**Figure 5**) and Dr. Carlos M. Cuevas (in Argentina) (**Figure 4**) directed this thesis in a very efficient and friendly way. Prof. Gatze Lettinga was the inspiring presence behind this research (**Figure 5**, **Figure 6**), which had to be finished in troubled times (**Figure 7**).

ALGEMENE SAMENVATTING[2]

Het kader

In de provincie Salta, in het noordwesten van Argentinië, is de veel voorkomende lozing van onbehandeld of slecht behandeld rioolwater in rivieren, meren, open kanalen en andere waterlopen, zoals in vele ontwikkelingslanden, één van de meest acute milieuproblemen. Dit leidt tot ernstige bedreigingen van de volksgezondheid, milieuproblemen en tast het landschap aan. De economische gevolgen zijn o.a. een vermindering van de waarde van bezittingen, het verlies van traditionele recreatieve activiteiten zoals vissen of kamperen en de vermindering van opbrengsten uit toerisme. Zoals gebruikelijk, worden hierbij de arme en minder bevoorrechte sectoren van de maatschappij het eerst, harder en soms onomkeerbaar geraakt. De maatregelen die normaal worden ondernomen om dit probleem aan te pakken zijn over het algemeen georiënteerd op de bouw van grote, dure en complexe (aërobe) zuiveringssystemen in sommige steden en slecht ontworpen en onderhouden stabilisatievijvers aan de rand van kleine dorpen en steden. Met een stijgende urbanisatie raakt het goedkope land dichtbij de steden op. Hierdoor ontstaat de behoefte om de stabilisatievijvers op aanzienlijke afstand van de bevolkingscentra te bouwen, hetgeen langere en duurdere rioolsystemen vereist. Bovendien staan de stabilisatievijvers onder aanzienlijke druk om zich op grotere afstand te vestigen wegens klachten over stankoverlast en de potentiële verspreiding van watergerelateerde ziekten.

Om de hierboven gegeven redenen hebben de ontwikkelingslanden een dringende behoefte aan eenvoudige, betaalbare, compacte, flexibele, en efficiënte systemen om het rioolwater te behandelen. De anaërobe behandelingstechnologie voldoet aan deze voorwaarden. Tot voor kort geloofde men dat rioolwater slechts anaëroob kan worden behandeld in tropische landen, waar de temperatuur van rioolwater over het algemeen boven de 25°C is. Het klimaat in Argentinië is over het algemeen gematigd en werd daarom ongeschikt geacht voor anaërobe rioolwaterzuivering. Bovendien werd de anaërobe behandeling van rioolwater slechts voordelig geacht indien de concentratie van het Chemische Zuurstof Verbruik (CZV) hoog was (hoger dan 500 mg CZV/L). Hierdoor werd de haalbaarheid van anaërobe behandeling van verdund rioolwater (minder dan 250 mg CZV/L), hetgeen in Argentinië eerder de norm dan de uitzondering is, betwijfeld.

Het eerste en belangrijkste technische obstakel voor de anaërobe behandeling van rioolwater in subtropische gebieden met “upflow anaerobic sludge bed” (UASB) reactoren, was de aanwezigheid van gesuspendeerde deeltjes (SS) in het water. De hydrolyse, de snelheidsbeperkende stap van het gehele proces van anaërobe vergisting van organische deeltjes, kan bij lage temperaturen te langzaam worden, wat weer kan leiden tot accumulatie van niet omgezette deeltjes in het slibbed van de UASB. Onder deze omstandigheden kan de slibverblijftijd (SRT) in de reactor te kort worden om een goede mate van zuivering, en een voldoende slibstabilisatie te realiseren bij een niet te lange hydraulische verblijftijd (HVT). De voorbehandeling voor de verwijdering van deeltjes kan onmisbaar worden wanneer het rioolwater in UASB reactoren bij korte HVT en lage temperatuur wordt behandeld. De toepassing van tweetraps anaërobe systemen werd al eerder gesuggereerd om de problemen m.b.t. de accumulatie van deeltjes onder deze omstandigheden te overwinnen.

[2] Translation: Patrick G. Todd and Grietje Zeeman.

Het werk

In hoofdstuk 1 wordt een uitvoerig literatuuroverzicht t.a.v. het gebruik van opstroom reactoren voor de behandeling van rioolwater gegeven. Hierbij worden experimenten uit de hele wereld op laboratoriumschaal, pilot schaal en in de praktijk beschreven en besproken.

Hoofdstuk 2 bevat een uitvoerige karakterisering van verschillende typen rioolwater van de stad Salta. De gemeten parameters omvatten de temperatuur, basissamenstelling (de pH, het organische stofgehalte, de alkaliniteit, de concentratie vluchtige vetzuren, het gehalte vaste stoffen en opgeloste stoffen, nutriënten, enz.), de anaërobe biologische afbreekbaarheid, en de hydrolyseconstante van gesuspendeerde organische deeltjes. Een volledige beschrijving van het betreffende rioolwater vormde een essentiële eerste stap in de beoordeling van de haalbaarheid van anaërobe behandeling van rioolwater in het gebied.

De resultaten beschreven in hoofdstuk 3 en 4 tonen de prestaties van een systeem dat bestaat uit een primaire sedimentatietank die wordt gevolgd door een experimentele UASB reactor. Dit was het eerste experiment in Argentinië op het gebied van anaërobe rioolwaterzuivering. Het systeem was operationeel gedurende meer dan zeven jaar in Salta, waar de gemiddelde omgevingstemperatuur 16,5ºC is en temperaturen onder nul gedurende enkele weken per jaar voorkomen (**Figuur 1**). Tijdens de experimenten, was de gemiddelde jaarlijkse temperatuur van het rioolwater 23,0ºC. De laagste gemiddelde dagelijkse temperatuur tijdens deze periode was 12.6ºC. Tijdens deze periode werd een zeer hoge verwijdering van het chemisch zuurstof verbruik (CZV) waargenomen, namelijk 84% voor totaal CZV en 92% voor gesuspendeerd CZV, bij een gemiddelde HVT van 2 uur in de sedimentatietank en 5,6 uur in de UASB reactor,overeenkomend met een opstroomsnelheid (V_{up}) van 0,71 m/uur. In de reactor ontwikkelden zich grote slib korrels, waarschijnlijk vanwege de lage concentratie aan gesuspendeerde deeltjes en CZV in het influent, in combinatie met een adequate HV en V_{up} (**Figuur 2**). De specifieke methanogene activiteit (SMA) van het slib was betrekkelijk laag (0,10 gCZV-CH_4/gVSS.d)[3] en waarschijnlijk bevatte het korrelslib (en dan vooral de grotere korrels) aanzienlijke hoeveelheden dode anaërobe bacteriën en inerte organische stof. De groeisnelheid van het korrelslib was zeer klein en berekeningen tonen aan dat slechts één lozing per twee jaar voldoende zou kunnen zijn om het spuislib te verwijderen. Het bestudeerde systeem was extreem robuust en efficiënt en leverde een constant effluent dat voldeed aan de geldende lozingsnormen voor CZV en deeltjes. Een veilige en efficiënte zuivering zou kunnen worden bereikt in een 4m hoge UASB reactor, wanneer het slibbed tussen de 1 en 2 meter hoog wordt gehouden en een opstroomsnelheid (Vup) van *ca.* 0,5 m/h (HVT ≈ 8 h) toegepast wordt.

Nadat de (technische) haalbaarheid van anaërobe behandeling van voorbezonken rioolwater onder de lokale omstandigheden was aangetoond, werd de directe anaërobe behandeling van ruw rioolwater getest. Hiervoor werd een tweetraps UASB systeem gebouwd en in werking gesteld (**Figuur 3**). De geraadpleegde literatuur wijst erop dat het gebruik van tweetraps anaërobe systemen, de slibretentie en de afbraak van gesuspendeerde deeltjes (SS) bij lage temperaturen verbeteren. De resultaten in hoofdstuk 5 toen aan dat een tweetraps anaëroob systeem zeer efficiënt en robuust is onder de toegepaste omstandigheden. Een totale verwijdering van 89% van het CZV werd verkregen in het tweetraps anaërobe systeem bij een HVT van 6,4 + 5,6 uur (V_{up} = 0,62 + 0,70 m/uur), met respectievelijk 83% en 36% verwijdering in de eerste en tweede stap. De kwaliteit van het effluent komt overeen met dat van de sedimentatietank gevolgd door een UASB reactor. Omdat

[3] Methaan productie uitgedrukt in gram COD per gram organische gesuspendeerde stof per dag.

de eerste anaërobe reactor het grootste gedeelte van het inkomende CZV verwijderde bij een zeer redelijke HVT, was de tweede stap vrijwel overbodig. Dit gegeven vormt een belangrijke wetenschappelijke doorbraak en betekent dat de directe anaërobe behandeling van rioolwater in UASB reactoren niet alleen onder tropische maar ook onder subtropische omstandigheden mogelijk en efficiënt is.

De prestaties van het systeem werden nauwelijks beïnvloed bij toepassing van lage temperaturen gedurende de winterperiode van ongeveer drie maanden. Het geproduceerde slib in de eerste reactor was goed gestabiliseerd (met name in de zomertijd) en kan zonder verdere behandeling worden gespuid. Een andere interessante waarneming was dat het opstarten van de eerste UASB reactor bij een HVT van 8,2 uur, en een opstroomsnelheid (V_{up}) van 0,48 m/uur slechts één maand duurde, terwijl geen entmateriaal was toegevoegd. De SMA in de eerste en tweede UASB reactor was respectievelijk 0,12 en 0,04 gCZV-CH_4/gVSS.d. De slibaanwas in de eerste UASB reactor was minder dan 5% van het reactorvolume per maand. Uit berekeningen volgt dat het overtollige slib slechts één of twee keer per jaar hoeft te worden gespuid, hetgeen verstoringen van de reactor als gevolg van menselijke fouten minimaliseert.

Rioolwater, dat alleen anaëroob wordt behandeld, produceert een effluent dat niet direct op het oppervlaktewater kan worden geloosd omdat de concentratie van o.a. ziekteverwekkers te hoog is. Nabehandeling van het effluent van de UASB reactoren is daarom noodzakelijk. Stabilisatievijvers vormen hiervoor een goedkope en efficiënte optie. Een systeem bestaande uit vijf stabilisatievijvers in serie, werd onderzocht voor de nabehandeling van het anaërobe effluent (**Figuur 3**). De vijvers werden alleen toegepast voor de verwijdering van pathogene micro-organismen.. In het systeem kon bovendien een propstroom regime worden benaderd, zodat een relatief klein oppervlakte kon worden toegepast. Zoals aangetoond in hoofdstuk 5, voldeed de concentratie aan ziekteverwekkers in het effluent van de stabilisatievijvers aan de strenge lozingseisen. De verhoging van de CZV concentratie in de vijvers, als gevolg van de groei van algen, resulteerde niet in een overschrijding van de lozingsnormen.. Als algemene conclusie, kan worden gesteld dat de UASB reactor gevolgd door een reeks van stabilisatievijvers een zeer efficiënt, betrouwbaar, en eenvoudig systeem is voor de behandeling van ruw rioolwater in subtropische gebieden, zoals Salta.

De resultaten, die in hoofdstukken 2 tot 5 zijn beschreven en besproken, tonen aan dat de bestudeerde systemen efficiënt en betrouwbaar zijn en dat praktijktoepassing kan worden gerealiseerd. Een zuiver wetenschappelijke beoordeling is echter ontoereikend om de acceptatie en de implementatie van deze technologie ter plaatse te garanderen. In hoofdstuk 6, wordt duidelijk gemaakt dat "duurzaamheid" een criterium kan zijn om verschillende (milieu) technologieën te beoordelen. Het gebruik van dit criterium als besluitvormingsinstrument heeft theoretische en morele wortels in het concept "duurzame ontwikkeling" dat in 1987 in het politieke programma van de World Commission on Environment and Development (WCED) werd geïntroduceerd. Een objectieve beoordeling van duurzaamheid zou rekening moeten houden met de mondiale, lange termijn context waarin de technologie wordt gebruikt. Er zijn namelijk geen of weinig technologieën die als intrinsiek duurzaam kunnen worden beschouwd. Als praktisch voorbeeld werd een vergelijkend onderzoek naar drie technologische opties voor de behandeling van rioolwater van de stad Salta uitgevoerd met behulp van een reeks technische, milieu, sociale, en economische criteria en indicatoren. De volgende technologische opties werden vergeleken: (A) een hoogbelast aëroob systeem (met primaire en secundaire sedimentatie, oxidatiebed, anaërobe afbraak van het slib en chlorering); (B) Drie conventionele stabilisatievijvers in serie; en (C) de UASB reactoren met nabehandeling in een kleine stabilisatievijver. De selectie van deze technologieën werd gebaseerd op de directe beschikbaarheid in het gebied. Uit deze eerste beoordeling blijkt dat, onder lokale omstandigheden, een systeem met een UASB en stabilisatievijver duurzamer is dan de andere twee onderzochte opties (hoofdstuk 6). De gebruikte methode was eenvoudig en gevoelig genoeg om verschillen in duurzaamheid tussen de vergeleken systemen te onderkennen. Deze methode moet echter niet als blauwdruk voor de beoordeling van duurzaamheid in het algemeen worden gezien. De selectie van de meest geschikte methode voor een specifieke situatie, hangt af van het type beoordeling dat dient te worden uitgevoerd en van vele andere plaatsspecifieke voorwaarden. In hoofdstuk 6 wordt de behoefte aan rationele, eenvoudige en objectieve methodes om de duurzaamheid van verschillende technologieën te beoordelen, benadrukt en besproken. Voordat een concrete beleidsbeslissing wordt genomen, is het noodzakelijk om een dergelijke beoordeling op een transparante en participatieve manier uit te laten voeren door een representatieve commissie van lokale actoren, stakeholders en deskundigen. De conclusie van hoofdstuk 6 en van dit proefschrift is dat duurzame ontwikkeling alleen kan worden verkregen door middel van volledig democratische technologie beoordeling en besluitvorming, welke boven politieke en economische motivaties staan en milieuproblemen en sociale onrechtvaardigheden kunnen erkennen en op lossen.

De mensen

Deze thesis, die onder de supervisie van Dr. Grietje Zeeman (**Figuur 5**) en Carlos M. Cuevas (**Figuur 4**) stond, werd dankzij de gezamenlijke inspanning van het Laboratorium van Milieustudies (Laboratorio De Estudios Ambientales, LEA) voltooid (**Figuur 4**), Prof. Gatze Lettinga vormde de inspirerende kracht achter dit onderzoek (**Figuur 5, Figuur 6**). Dit alles zorgde ervoor dat dit onderzoek ondanks de moeilijke omstandigheden succesvol werd voltooid (**Figuur 7**).

RESUMEN GENERAL

El marco

En la provincia de Salta, en el noroeste de Argentina, así como en muchos países en desarrollo, uno de los problemas ambientales más acuciantes es la descarga de líquidos cloacales crudos o pobremente purificados en ríos, lagos, canales y otros cursos de agua. Esta práctica trae aparejadas serias amenazas a la salud pública, agudos problemas ambientales y deterioro creciente del paisaje. Entre las consecuencias económicas pueden citarse, entre otras, la reducción en el valor de la propiedad, la pérdida de actividades recreativas tradicionales tales como la pesca o el camping y la virtual desaparición de ingresos provenientes del turismo. Como es norma en estos casos, los sectores más pobres y menos privilegiados de la sociedad civil son los primeros en ser afectados, muchas veces de manera irreversible. Las acciones emprendidas por los gobiernos para atacar este problema se han orientado generalmente hacia la construcción de grandes, caras y complejas plantas de tratamiento de tipo aeróbico en algunas ciudades, o de sistemas de lagunas de estabilización (LDE), a menudo mal diseñadas y mantenidas, en las afueras de ciudades más pequeñas o pueblos. A medida que el fenómeno de la urbanización avanza, los terrenos cercanos a las ciudades son cada vez más caros y las LDE deben ser construidas a mayores distancias, lo cual también requiere sistemas de cloacas (o alcantarillado) más prolongados. Algunas LDE en funcionamiento están siendo fuertemente cuestionadas por problemas de emanación de olores y a causa de los potenciales riesgos de transmisión de enfermedades vehiculizadas por el agua. Estas LDE probablemente deban ser relocalizadas a mayor distancia de los centros urbanos.

Por las razones expuestas arriba, los países en desarrollo necesitan con urgencia sistemas de tratamiento de líquidos cloacales que sean simples, accesibles en términos de costos, compactos, flexibles y eficientes. La tecnología anaeróbica parece cumplir bastante bien con estos requisitos. Sin embargo, se creyó hasta muy recientemente que los líquidos cloacales solo podían ser tratados de manera anaeróbica en países tropicales, donde la temperatura de los líquidos cloacales está generalmente por encima de 25°C. El clima en Argentina es mayormente templado y, por lo tanto, se suponía que el tratamiento anaeróbico de líquidos cloacales estaba vedado para este país. Por otra parte, el tratamiento anaeróbico sólo se consideraba ventajoso para altas concentraciones de demanda química de oxígeno (DQO) en los líquidos cloacales (más de 500 mg/L). Por lo tanto, la factibilidad del tratamiento anaeróbico era fuertemente cuestionada para los tipos de líquidos cloacales diluidos (menos de 250 mgDQO/L) que parecen ser más la norma que la excepción en Argentina.

El primer y principal obstáculo técnico para el tratamiento de líquidos cloacales en regiones subtropicales usando reactores de flujo ascendente y manto de lodos (o reactores UASB, sigla en inglés de “upflow anaerobic sludge bed”) era supuestamente la presencia de sólidos suspendidos (SS). La hidrólisis, el paso limitante en el proceso de digestión anaeróbica de materia orgánica particulada, puede resultar demasiado lento a bajas temperaturas, lo que conduce a una acumulación de SS sin degradar en el manto de lodos del reactor. Bajo estas circunstancias, el tiempo de retención de sólidos (TRS) en el reactor puede resultar demasiado corto para proveer una buena eficiencia de tratamiento y una satisfactoria estabilización de lodos a un tiempo de retención hidráulico (TRH) razonable. Por lo tanto, se requiere la remoción previa de los SS cuando se deben tratar líquidos cloacales a bajos TRH y bajas temperaturas. También se ha indicado que el problema de la acumulación de SS bajo estas condiciones operativas podría resolverse mediante el uso de sistemas anaeróbicos en dos etapas.

El trabajo

En el capítulo 1 se presenta una revisión completa de la bibliografía existente a la fecha, referida al uso de reactores de flujo ascendente para el tratamiento de líquidos cloacales. Se describen y discuten experimentos y aplicaciones de laboratorio, de planta piloto y a escala real en todo el mundo. También se identifican algunas áreas que requerirían investigación posterior.

En el capítulo 2 se describe la exhaustiva caracterización física, química y biológica realizada a distintos tipos de líquidos cloacales de la ciudad de Salta. Entre los parámetros medidos se incluyó la temperatura, la composición básica (pH, materia orgánica, alcalinidad, ácidos grasos volátiles, sólidos totales y suspendidos, nutrientes, etc.), la biodegradabilidad anaeróbica y la constante de hidrólisis de la materia orgánica particulada. Una descripción completa de los líquidos cloacales se consideró indispensable como primer paso en la evaluación de la factibilidad del tratamiento anaeróbico en la región.

Los resultados presentados en los capítulos 3 y 4 muestran el rendimiento de un sistema de tratamiento de líquidos cloacales que consiste en un tanque de sedimentación primaria (sedimentador) seguido de un reactor UASB a escala piloto. El sistema fue operado por más de siete años en la ciudad de Salta, donde la temperatura media del ambiente es 16.5°C, registrándose temperaturas inferiores a 0°C duante varias semanas por año (**Figura 1**). Durante los experimentos, la temperatura media anual del líquido cloacal fue 23.0°C. Se observaron altas eficiencias de remoción de DQO (hasta 84% en DQO total y 92% en DQO suspendida) en el sistema, a un TRH promedio de 2 h en el sedimentador y 5.6 h en el reactor, equivalente a una velocidad ascensional (V_{up}) de 0.71 m/h. Se observó el desarrollo de gránulos de gran tamaño en el reactor, probablemente debido a las bajas concentraciones de SS y DQO en el influente y a una adecuada combinación de TRH y V_{up} (**Figura 2**). La actividad metanogénica específica (AME) del lodo fue relativamente baja (0.10 gDQO-CH_4/gSSV.d)[4] y probablemente la mayoría de los gránulos (especialmente los más grandes) estaban formados por cantidades considerables de bacterias anaeróbicas muertas y materia orgánica inerte. El crecimiento del manto de lodo fue muy lento, y se calcula que una descarga cada dos años podría ser suficiente para la eliminación del exceso de lodo. El sistema estudiado fue altamente robusto y eficiente, y produjo de manera constante un efluente que cumplía con los límites establecidos de descarga para DQO y SS. En reactores típicos de 4 m de altura, se podría alcanzar una operación segura y eficiente si se mantiene el manto de lodo a una altua de entre 1 y 3 m, y se aplica una V_{up} de alrededor de 0.5 m/h (TRH $\approx$ 8 h).

Habiendo demostrado la factibilidad técnica, bajo las condiciones locales, de la aplicación de sistemas anaeróbicos para el tratamiento de líquido cloacal sedimentado, se estudió su aplicación para el tratamiento de líquido cloacal crudo. Para ello, se construyó y operó un sistema de reactores UASB de dos etapas (**Figura 3**). En el capítulo 5 se demuestra que un sistema anaeróbico de 2 etapas es también altamente eficiente y robusto. Se obtuvo una eficiencia de remoción de DQO total de 89% en las dos etapas a TRH de 6.4 + 5.6 h (V_{up} = 0.62 + 0.70 m/h), con 83 y 36% de remoción en la primera y segunda etapa, respectivamente. La concentración del efluente fue similar a la obtenida en el sistema Sedimentador + reactor UASB descripto en el capítulo 3. Además, como el

[4] Producción de metano, en gramos de DQO, por gramo de SS volátiles y por día.

primer reactor removió la mayor parte de la DQO entrante a un TRH muy razonable, la segunda etapa se consideró redundante en la práctica. Esto se considera un avance significativo y significa que el tratamiento directo de líquido cloacal crudo es posible (y, de hecho, eficiente) bajo condiciones subtropicales en reactores UASB de una sola etapa, tal como ocurre en regiones tropicales. El rendimiento del sistema no estuvo afectado durante la etapa mas fría del año, que generalmente dura unos tres meses. El lodo retenido en el primer reactor presentó una buena estabilidad, especialmente durante el verano, y podría ser directamente eliminado sin ningún tipo de tratamiento ulterior. Otra observación interesante fue que la puesta en marcha del primer reactor UASB fue conseguida en sólo un mes, sin el agregado de inóculo de ningún tipo, a un TRH de 8.2 h y una V_{up} de 0.48 m/h. La AME en el primer y segundo reactor fue 0.12 y 0.04 gDQO-CH_4/gSSV.d, respectivamente. La tasa de crecimiento del lodo en el primer reactor fue menos de 5% del reactor (en volumen) por mes. Se calcula que el lodo en exceso podría ser eliminado mediante sólo una o dos descargas por año, con lo cual se minimizarían tanto la alteración del proceso biológico en el reactor como los posibles errores humanos durante la operación.

El tratamiento anaeróbico por sí solo no consigue proveer un efluente que pueda ser descargado en un río o cuerpo de agua, principalmente porque la concentración de microorganismos patógenos es demasiado alta. Por tal motivo, cuando los reactores UASB se usan para el tratamiento de líquidos cloacales, generalmente se necesita una etapa de post-tratamiento. Las LDE son una opción barata y eficiente para este propósito. En este trabajo se estudió un sistema de cinco LDE en serie como sistema de post-tratamiento para las etapas anaeróbicas (**Figura 3**). Las lagunas solamente eran necesarias para la remoción de patógenos y, por lo tanto, el área superficial necesaria de diseño fue relativamente pequeña, sobre todo porque, con cinco unidades en serie, es posible aproximarse razonablemente a un modelo hidráulico de flujo pistón. Como se muestra en el capítulo 5, la concentración de patógenos en el efluente final de las lagunas cumplió con las más exigentes normas de volcamiento. El incremento observado en la concentración de DQO en las lagunas (debido al crecimiento de algas) no fue motivo suficiente para que el efluente exceda el límite admitido para DQO. Como conclusión general de esta parte del trabajo, se puede afirmar que un reactor UASB seguido por LDE en serie es un sistema eficiente, compacto, confiable y simple para el tratamiento de líquidos cloacales crudos en regiones subtropicales como Salta.

Los resultados presentados y discutidos en los capítulos 2 al 5 mostraron que los sistemas estudiados fueron eficientes y confiables, y que su aplicación a escala real en la región es altamente factible. Sin embargo, una evaluación puramente científica se considera insuficiente para garantizar la aceptación y la utilización de esta tecnología. En el capítulo 6 se sostiene que la noción de "sustentabilidad" puede ser un criterio amplio para la evaluación de diferentes tecnologías ambientales. El uso de este criterio como una herramienta para la toma de decisiones tiene raíces teóricas y morales en el concepto de "desarrollo sustentable" introducido en la agenda política en 1987 por la Comisión Mundial sobre Ambiente y Desarrollo (conocida como WCED, sigla en inglés de "World Commission on Environment and Development"). Una evaluación de sustentabilidad amplia debería tomar en cuenta el contexto global y de largo plazo en el que la tecnología es usada, porque se considera que no hay tecnologías que puedan ser consideradas intrínsecamente sustentables (o hay muy pocas). Como ejemplo práctico, se llevó a cabo una evaluación comparativa de tres opciones tecnológicas para el tratamiento de líquidos cloacales para la ciudad de Salta, en términos de una serie de criterios e indicadores de carácter técnico, ambiental, social, y económico. Las opciones comparadas fueron las siguientes: (A) sistema de tratamiento aeróbico de alto rendimiento (sedimentadores primarios y secundarios, lechos percoladores, digestión de lodos y cloración); (B) tres LDE convencionales en serie; y (C) reactor UASB con post-tratamiento en pequeñas LDE. La selección de estas tecnologías se basó en su inmediata disponibilidad en la región. En esta evaluación preliminar se encontró que, bajo las condiciones locales, un sistema UASB + LDE es más sustentable que las otras dos opciones (los resultados se muestran en el capítulo 6). El método de evaluación fue simple de aplicar y sensible para detectar diferencias de sustentabilidad entre las diferentes opciones comparadas. Sin embargo, este método no debe considerarse como una receta para la evaluación de la sustentabilidad. La selección del método más apropiado para cada caso dependerá del tipo de evaluación que deba realizarse y de muchas otras condiciones que dependen en gran medida de las condiciones particulares del lugar de realización. En el capítulo 6 se enfatiza y discute la necesidad de contar con un método racional, simple y abarcativo para la evaluación de la sustentabilidad de diferentes tecnologías. También se subraya que, antes de tomar una decisión concreta, la evaluación definitiva debe hacerse a través de un panel de actores locales, expertos y otros interesados, de forma transparente y representativa. El capítulo 6 concluye que una manera completamente democrática de toma de decisiones y evaluación de tecnología que pueda trascender las meras motivaciones políticas y económicas y que pueda reconocer, tener en cuenta y resolver los problemas ambientales y las injusticias sociales es probablemente el único camino para alcanzar el desarrollo sustentable.

La gente

Esta tesis no podría haberse realizado sin el esfuerzo coordinado llevado a cabo por los integrantes del Laboratorio de Estudios Ambientales (LEA) (**Figura 4**). La Dra. Grietje Zeeman (en Holanda) (**Figura 5**) y el Dr. Carlos M. Cuevas (en Argentina) (**Figura 4**) dirigieron esta tesis de una manera amigable y eficiente a la vez. La presencia inspiradora del Prof. Gatze Lettinga estuvo siempre detrás de esta investigación (**Figura 5, Figura 6**), la cual debió ser finalizada en tiempos turbulentos (**Figura 7**).

Figure 1. Pilot plant 1. Settler (not shown) and UASB reactor (white tank) on a background of “subtropical” snow.
Figuur 1. Proefinstallatie 1, sedimentatietank (niet getoond) en UASB reactor (witte tank) op een achtergrond van “subtropisch” sneeuw.
Figura 1. Planta piloto 1. Sedimentador primario (no en la foto) y reactor UASB (tanque blanco) sobre un fondo de nieve “subtropical”.

Figure 2. Granules observed in the UASB reactor fed with settled sewage.
Figuur 2. Korrels die in de reactor UASB worden waargenomen die met het bezonken rioolwater wordt gevoed.
Figura 2. Gránulos observados en el reactor UASB alimentado con líquido cloacal sedimentado.

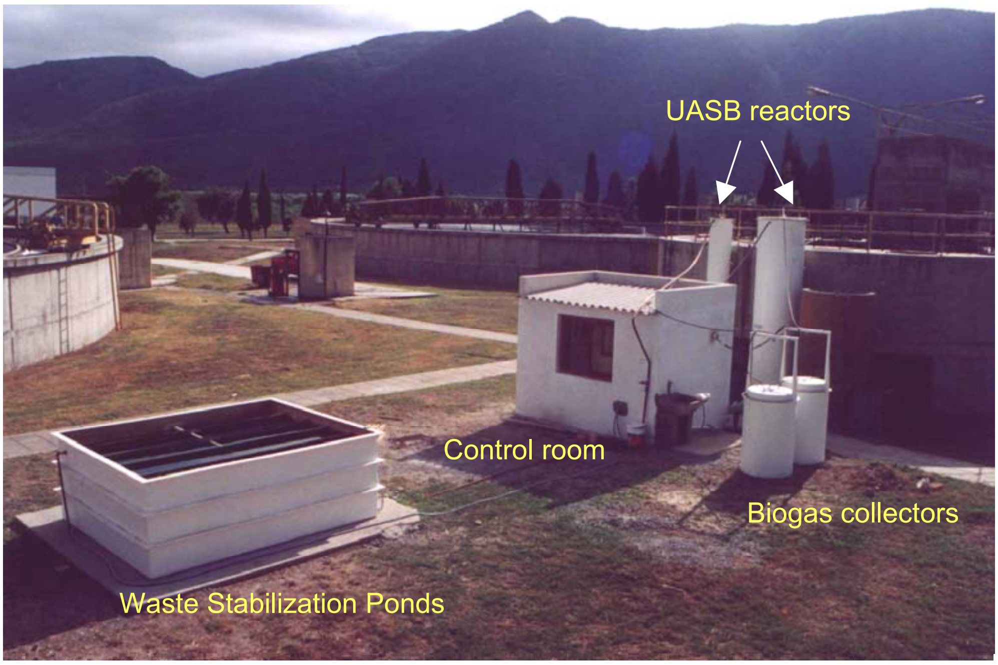


Figure 3. Pilot plant 2. Integrated treatment of raw sewage.
Figuur 3. Proefinstallatie 2. Geïntegreerde behandeling van ruw rioolwater.
Figura 3. Planta piloto 2. Tratamiento integral de líquido cloacal crudo.

Figure 4. Some of the members of the “Laboratorio de Estudios Ambientales”.
Figuur 4. Enkele leden van "Laboratorio de Estudios Ambientales".
Figura 4. Algunos de los miembros del “Laboratorio de Estudios Ambientales”.
Standing from left to right/ Staand van links naar rechts/Parados de izquierda a derecha: Aníbal, Carlos (“el Doctor”), Martín, Jimena, Estela, Walter, Viviana, Alejandra, Ana María. Sitting from left to right/ Zittend van links naar rechts/Sentados de izquierda a derecha: the author/de auteur/el autor, Marcelo, Ana, Julio, Carolina, Silvia. Not in the picture/ Niet afgebeeld/No en la foto: María Laura, Raquel, Patrick.

Figure 5. Gatze Lettinga and Grietje Zeeman in Wageningen (November 1999).
Figuur 5. Gatze Lettinga en Grietje Zeeman in Wageningen (November 1999).
Figura 5. Gatze Lettinga y Grietje Zeeman en Wageningen (Noviembre 1999).

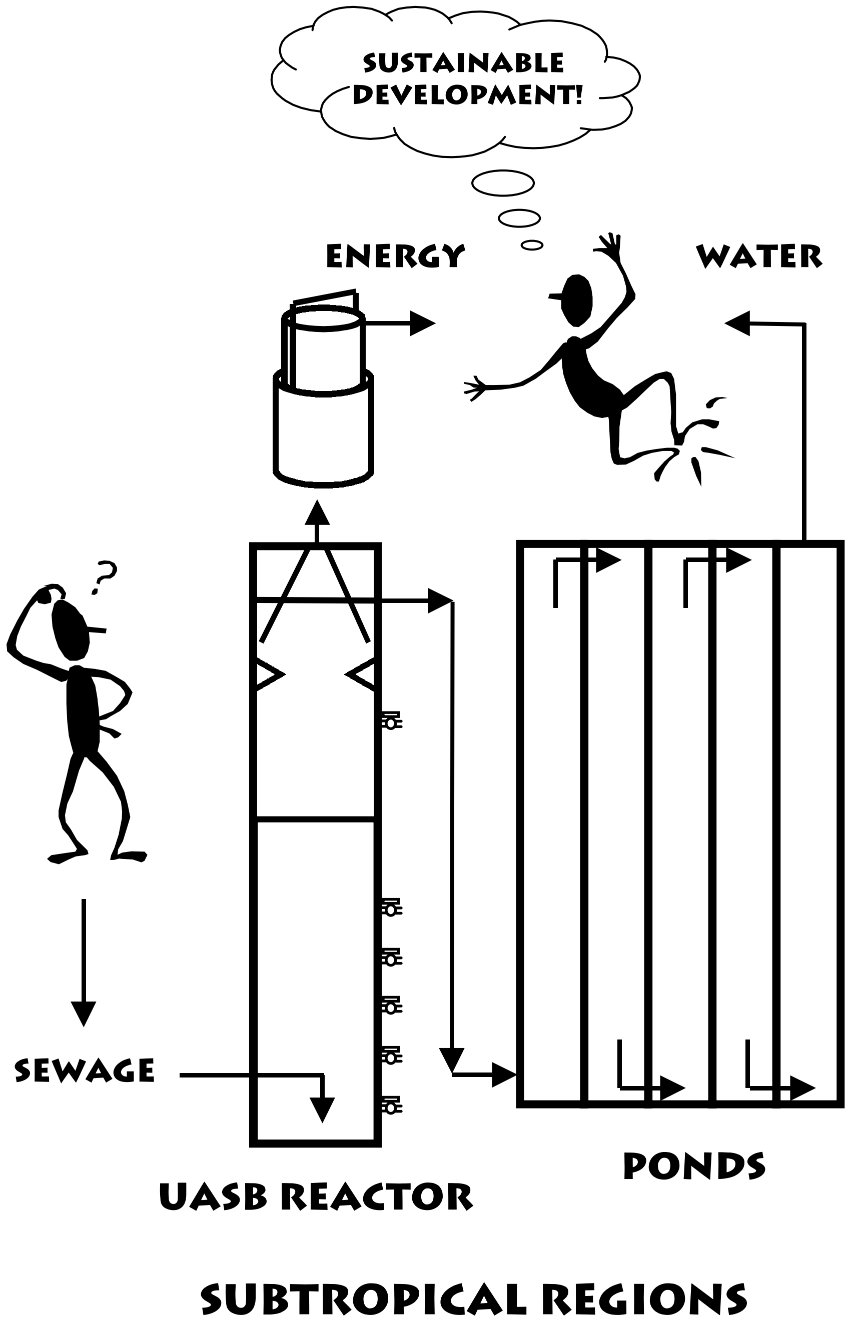
Figure 6. Gatze Lettinga and the author in pilot plant 1 (November 2000).
Figuur 6. Gatze Lettinga en de auteur bij de proefinstallatie 1 (November 2000).
Figura 6. Gatze Lettinga con el autor en la planta piloto 1 (Noviembre 2000).

Figure 7. Street protests in Salta during the economic and social crisis that struck the country at the end of 2001. From left to right: Lucio, Patrick, and the author trying to make the best out of a difficult time.

Figuur 7. De straatdemonstraties in Salta tijdens de economische en sociale crisis aan het eind van 2001. Van links naar rechts: Lucio, Patrick, en de auteur die het beste uit moeilijke tijden proberen te maken.

Figura 7. Protestas callejeras en Salta durante la crisis económica y social que sacudió al país a finales del 2001. De izquierda a derecha: Lucio, Patrick y el autor tratando de pasarla lo mejor posible en tiempos difíciles.

GRAPHIC MINI-SUMMARY



CURRICULUM VITAE

The author of this dissertation, Lucas Seghezzo, was born on April 8th, 1965, in Carlos Paz, province of Córdoba, Argentina. He started his university studies in Natural Resources in 1983 at the National University of Salta (UNSa), Argentina, and he obtained his degree in 1990. His graduate thesis was on the production of biogas in greenhouses. After that, he did research on anaerobic digestion and wastewater treatment for about five years with fellowships from the National Research Council of Scientific and Technical Research (CONICET). He is married since 1992 to Adriana Alvarez and he has two children: Natalia (6) and Mateo (3). In 1995 he obtained a fellowship from the Netherlands Organization for International Cooperation in Higher Education (NUFFIC) to do a M.Sc. on Environmental Sciences, specialization Environmental Technology, in Wageningen University, the Netherlands. He graduated with distinction in 1997 with a thesis on kinetics and mass transfer phenomena in anaerobic granular sludge. Back in Argentina, he was appointed Director of Environmental Protection at the Municipality of Salta during two years. After that, he worked two more years as environmental advisor to the Municipal Legislative Council. From 1998 to 2002, he was also part-time lecturer of “Environmental Sanitation” at the National University of Salta. As a private consultant he coordinated several environmental impact assessments, environmental auditing programs, and environmental monitoring studies. He also designed and constructed small-scale integrated sewage treatment systems for companies, small private neighborhoods, and single houses. His “sandwich” Ph.D. program started officially in 1997 at the Sub-Department of Environmental Technology of Wageningen University.

Current address:

Treubstraat 77
6702 BB Wageningen
The Netherlands
Phone: +31-(0)317-840350
Email: Lucas.Seghezzo@wur.nl; Lucas_Seghezzo@hotmail.com

Address in Argentina:

Universidad Nacional de Salta
Consejo de Investigación-INENCO
Laboratorio de Estudios Ambientales
Buenos Aires 177
A 4402 FDC Salta
Argentina
Phone: +54-(0)387-4255516
Fax: +54-(0)387-4255483
Email: lucas@unsa.edu.ar

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