**KINETIC EVALUATION OF THE PSYCHROPHYLIC ANAEROBIC
DIGESTION OF SYNTHETIC DOMESTIC SEWAGE USING AN UPFLOW
FILTER**

M.A. Martín¹, M.A. De la Rubia², A. Martín¹, R. Borja^{2*}, S. Montalvo³, E. Sánchez⁴

¹ Departamento de Química Inorgánica e Ingeniería Química, Facultad de Ciencias,
Universidad de Córdoba, Campus Universitario de Rabanales, Edificio C-3, Ctra.
Madrid-Cádiz, km. 396, 14071-Córdoba, Spain.

² Instituto de la Grasa (C.S.I.C.), Avda. Padre García Tejero, 4, 41012-Sevilla, Spain.

³ Departamento de Ingeniería Química, Universidad de Santiago de Chile, Avenida
Libertador Bernardo O'Higgins 3363, Santiago de Chile, Chile.

⁴ Centro de Ciencias Medioambientales (C.S.I.C.), C. Serrano 115 duplicado, 28006,
Madrid, Spain.

* Corresponding author: R. Borja, Instituto de la Grasa (CSIC), Avda. Padre García
Tejero, 4. 41012-Sevilla, Spain (Tel.: +34-954-689-654; Fax: +34-954-691262; E-mail
address: rborja@cica.es).

ABSTRACT

A study of the anaerobic digestion process of synthetic domestic sewage (total COD: 705 mg/L) was carried out. The digestion was conducted in an upflow anaerobic filter with corrugated plastic rings as packing media at psychrophilic temperature (15°C-17°C). For HRTs of between 10.0 and 17.1 h, the total COD removal efficiency was almost constant and independent on the HRT, achieving an average value of around 80%. However, when the HRT decreased from 7.0 to 3.2 h the efficiency diminished from 77% to 65%. This decrease in removal efficiency was parallel to the increase in the VFA/Alkalinity ratio for this HRT range. The flow pattern observed in the reactor studied was intermediate between plug-flow and CSTR systems, although the plug-flow was predominant. It can also be observed that Young and McCarty's model almost coincided with the CSTR model, when the biodegradable COD was used for fitting the data. The Manariotis equation allowed a better fit of the experimental data (total COD removal efficiency with influent substrate concentration and HRT) than the Young model. The methane yield coefficient obtained was 0.15 L CH₄/g COD consumed.

Keywords: CSTR flow pattern; Domestic sewage; Manariotis equation; Plug-flow pattern; Upflow anaerobic filter; Young equation.

1. INTRODUCTION

Anaerobic purification of domestic sewage is a useful treatment system for small communities (Alderman et al., 1998; Barber and Stuckey, 1999; Bodik et al., 2002).

Compared to conventional aerobic processes, anaerobic wastewater treatment of domestic wastewater can be used as a viable alternative (Aiyuk et al., 2006). It produces only small amounts of stabilised (non-putrescible) sludge compared to aerobic processes, while much of the removed organic material is converted to methane. Furthermore, the need to aerate the wastewater is dispensed with, reducing energy and construction costs (Lettinga et al., 1997; Zakkour et al., 2001; Cakir and Stenstrom, 2005). Anaerobic processes have gained popularity over the past decade, and have already been applied successfully for the treatment of a number of waste streams, and geared mainly towards highly concentrated soluble wastewaters (Aiyuk et al., 2006). Scepticism relates to their applicability for low-strength wastewaters as domestic wastewater has been widespread for a considerable period. Recently, however, more efficient anaerobic systems have been developed, and they are being successfully applied for treatment of low-strength wastewaters such as domestic wastewater, particularly under tropical conditions where artificial heating can be avoided, to cut down on costs (Harleman and Murcott, 2001; Aiyuk and Verstraete, 2004).

Anaerobic filters have grown to represent advanced technology that has been used effectively for treating a variety of industrial wastewaters (Manariotis and Grigoropoulos, 2008). The upflow anaerobic filter is basically a contact process in which waste passes over or through a mass of biological solids contained within the reactor by a fixed media. The biomass in the reactor is attached to the medium surfaces as a thin biofilm, is entrapped within the media matrix, or is held as a granulated or flocculated sludge mass beneath the media. Soluble organic compounds in the influent wastewater pass in close proximity to this biomass and diffuse into the surfaces of the attached or granulated solids where they are converted to intermediate and end products, specifically methane and carbon dioxide (Young, 1991).

77 Compared with other high-rate anaerobic reactors, the anaerobic filter presents
78 several important characteristics (Manariotis and Grigoropoulos, 2008; Umaña et al.,
79 2008; Nikolaeva et al., 2009). The anaerobic filter is more suitable for handling high
80 pollution-load wastewaters because it has a high substrate removal efficiency.
81 Moreover, this type of reactor is less sensitive to shock loads and operates at lower
82 hydraulic retention times (HRTs), thus requiring smaller volumes. Construction,
83 operation and maintenance costs are lower. The effluent contains few suspended solids,
84 eliminating the need for the separation of solids or recycling and the biological system
85 recovers more quickly to the conditions present before the digester operation is stopped.
86 Therefore, non-feeding intervals do not have a negative impact on the quick response of
87 anaerobic filters (Manariotis and Grigoropoulos, 2008). These characteristics render the
88 anaerobic filter extremely useful for the treatment of both high and low-strength
89 wastewaters (Manariotis and Grigoropoulos, 2003).

90 Moreover, the anaerobic filter is recently gaining increased attention as an
91 alternative means for the direct anaerobic treatment of municipal and other low-strength
92 wastewaters at ambient conditions, especially in small decentralized facilities located in
93 regions of moderate climate (Manariotis and Grigoropoulos, 2006). Previous research
94 (Bodik et al., 2002) showed the effect of temperature (in the range 9°C-23°C) and HRT
95 (in the range 6 h-45 h) on the steady-state performance of a laboratory-scale anaerobic
96 filter filled with small tubes of plastic material treating a mixture of synthetic substrate
97 (glucose and sodium acetate) and real municipal wastewater with a COD of about 300
98 mg/L. COD removal efficiencies in the range of 46%-92% were obtained depending on
99 the temperature and HRTs used. Based on these results, the use of the anaerobic filter in
100 practice seems to be a potential technology for pre-treatment of sewage produced by
101 small communities. Another study of anaerobic digestion of low strength domestic

wastewater (COD: 288 mg/L) carried out with an anaerobic filter operating at 25°C and 35°C and at HRT of 24 h provided an average COD removal efficiency of 73% (Kobayashi et al., 1983).

Depending upon the temperature at which the process is carried out, there are three main types of anaerobic treatment of wastewaters. Biomethanation carried out at a temperature range of 45°C-60°C is referred to as “thermophilic”, whereas that carried out at a temperature range of 20°C-45°C is known as “mesophilic”. The anaerobic digestion of organic matter at low temperatures (< 20°C) is referred to as “psychrophilic” digestion (Kashyap et al., 2003). The biomethanation process at mesophilic and thermophilic ranges is well understood and documented. However, there is gap in the knowledge about the anaerobic digestion process at psychrophilic temperature. The details of the low temperature degradation pathways of diverse substrates and the microbial communities composition in various methanogenic environments are still not completely known (Bardiya and Chaudhari, 2000; Borja et al., 2002).

The aim of this work was to carry out a performance evaluation and kinetic study of the anaerobic digestion of a synthetic domestic sewage at psychrophilic temperature (15°C-17°C) by using an upflow filter operating at HRTs in the range of 3 h-12 h and organic loading rates in the range of 1-4 g COD/(L d). Different kinetic models that correlated the process efficiency with HRT and substrate concentration were tested and compared.

2. MATERIALS AND METHODS

2.1. Equipment

The anaerobic filter consisted basically of an acrylic column with a total volume of 1.5 L and an effective working volume of 1.35 L. The reactor column itself has a height of 45 cm and an internal diameter of 8 cm. 330 corrugated plastic rings with an individual diameter of 1.5 cm and a height of 0.5 cm were used as biomass growth support material. Effluent was recycled from the upper part to the bottom of the reactor to provide good mass transfer conditions. The reactor was fed by means of a peristaltic pump, and liquid effluent was removed continuously through a hydraulic seal designed to prevent air from entering the reactor and biogas from leaving it.

The methane volume produced in the process was measured using a 3-L Mariotte reservoir fitted to the reactor. A tightly closed bubbler containing a NaOH solution 3 M (with alizarin yellow as a CO₂ saturation indicator) to collect the CO₂ produced was intercalated between the two elements. The methane produced displaced a given volume of water from the reservoir, allowing to easily determine the methane generated. The operating temperature of the reactor (15°C-17°C) was maintained as constant by means of an external water jacket through which water from a temperature controlled bath circulated.

2.2. Synthetic domestic wastewater used

The composition of the synthetic domestic wastewater used is summarized in Table 1. The main characteristics and features of the wastewater used are shown in Table 2. This Table also shows the range of values of the typical parameters of a real domestic wastewater reported in the literature for comparison purposes (Kobayashi et al., 1983; Elmitwalli et al., 2002).

2.3. *Inoculum*

The reactor was inoculated with methanogenically active biomass from a laboratory-scale anaerobic reactor that operated at psychrophilic temperatures (15°C). Its content in total suspended solids (TSS) and volatile suspended solids (VSS) was 68.9 and 51.2 g/L, respectively. Finally, the pH of this anaerobic biomass was 7.2, while its methanogenic activity was 0.95 g COD-CH₄/(g VSS·d).

2.4. *Experimental procedure*

The anaerobic reactor was initially charged with 700 mL of the inoculum, 250 mL of distilled water, 300 mL of a nutrient-trace element solution and with 330 of the above-mentioned corrugated plastic rings. The composition of the nutrient-trace element solution used at the start-up of the reactor can be found elsewhere (Borja et al., 2001).

The start-up of the reactor involved gradual increases in COD loading. During this period the organic loading rate (OLR) was gradually increased from 0.1 to 0.3 g COD_T/(L·d) between days 1 and 15, 0.4 g COD_T/(L·d) between days 16 and 30, 0.6 g COD_T/(L·d) between days 31 and 45, and finally, 0.8 g COD_T/(L·d) between 46 and 60 days. During this start-up period, the superficial velocity was maintained at 0.1 m/h.

This gradual start-up process was followed by 16 series of continuous experiments using OLRs of 0.99, 1.15, 1.22, 1.26, 1.37, 1.42, 1.60, 1.71, 2.01, 2.26, 2.44, 3.00, 3.31, 4.05, 4.59 and 5.23 g COD_T/(L·d), which corresponded to HRTs of 17.1, 15.0, 14.0, 13.5, 12.5, 12.0, 10.5, 10.0, 8.7, 7.5, 7.0, 5.5, 5.0, 4.1, 3.6 and 3.2 h. During all these experiments the superficial velocity was maintained at 0.3 m/h with the recirculation of the effluent.

Once steady-state conditions were achieved at each OLR or HRT assayed, the daily volume of methane produced, total and soluble COD, pH, total volatile fatty acids

(VFA), and alkalinity of the different effluents obtained were determined. The samples were collected and analyzed for at least five consecutive days. The steady-state value of a given parameter was taken as the average of these consecutive measurements for that parameter when the deviations between the observed values were less than 5% in all cases. Each experiment had a duration of 6-7 times the corresponding HRT.

The OLRs applied in this study were increased in a stepwise fashion in order to minimize the transient impact on the reactor that might be induced by a sudden increase in loadings.

2.5. Chemical analyses

The following parameters were analyzed according to Standard Methods (APHA, 1998): total and soluble COD, pH, total solids (TS), mineral solids (MS), volatile solids (VS), total suspended solids (TSS), mineral suspended solids (MSS), volatile suspended solids (VSS), total volatile fatty acids (VFA), and alkalinity.

3. RESULTS AND DISCUSSION

3.1. Process stability

Table 3 summarizes the data of the effluent pH and VFA/Alkalinity ratio for all experiments carried out. As can be seen for HRTs of between 5.0 and 17.1 h, the pH in the reactor ranged between 7.30 and 7.45 showing that the buffering capacity of the experimental system was found to be at favourable levels. Only for a HRT of 3.2 h did the pH decrease to 6.95, although the methanogenic process was not seriously affected. This high stability can be attributed to carbonate/bicarbonate buffering. This is

produced by the generation of CO₂ in the digestion process, which is not completely removed from the reactor as gas. Buffering in anaerobic digestion is normally due to bicarbonate, as carbonate is generally negligible if compared with bicarbonate (carbonate/bicarbonate ratio is equal to 0.01 for pH 8.2) (Gujer and Zehnder, 1983; Speece, 1983).

On the other hand, the VFA/Alkalinity ratio can be used as a measure of process stability (Wheatley, 1990; Switzembaum et al., 1990): when this ratio is less than 0.5-0.6 the process is considered to be operating favourably without the risk of acidification. As was observed in Table 3, for HRTs higher than 3.6 h the ratio values were always lower than the suggested limit value, proving that the process is adequately stable. However, at a HRT of 3.2 h an increase in the VFA/Alkalinity ratio to a value of 0.51 was observed, which brought about a decrease in pH to 6.95 and the start of a slight destabilization.

3.2. Influence of the HRT on the process efficiency (E) and correlation of results

Figure 1 shows the variation of the total COD removal efficiency (per unit) with the HRT. As can be seen for HRTs of between 10.0 and 17.1 h, the removal efficiency was virtually constant and independent, achieving an average value of around 80%. However, when the HRT decreased from 7.0 to 3.2 h the efficiency diminished from 77% to 65%. This decrease in removal efficiency was parallel to the increase in the VFA/Alkalinity ratio for this HRT range (Table 3). Experiments with HRTs lower than 3.2 h were not performed because of the risk of reactor acidification. Therefore, for HRTs of between 10.0 and 17.1 h the reactor effluents achieved COD levels within regulation standards for treated effluent disposal. However, for HRTs lower than 5.5 h

some post-treatments could be necessary for achieving the mentioned standards (Diamadopoulos et al., 2007).

The COD removal efficiencies obtained in the present research (80% at HRTs in the range of 17.1 h-10.0 h) were higher than those obtained in another lab-scale anaerobic filter with tubes of PVC as packing material treating low strength municipal wastewater (efficiencies of 73%-77%) operating at higher HRT (24 h) and temperatures (20°C-25°C) (Kobayashi et al., 1983). These efficiency values were also higher than those obtained in lab-scale UASB and AnSBR reactors treating domestic wastewater, which achieved values of 64%-72% and 62% operating at temperatures of 13°C-19°C and 15°C, respectively, and HRTs of 5 h and 10 h-20 h, respectively (Uemura and Harada, 2000; Bodik et al., 2002). On the other hand, the efficiency values of the present study were of the same order of magnitude as those obtained in another lab-scale anaerobic filter treating a similar wastewater with plastic material as packing medium (84%) operating at a HRT of 10 h and 15°C temperature (Bodik et al., 2002). This study also revealed that a temperature higher than 10°C-12°C, COD removal efficiency was relatively independent on HRT in the range of 10 h-20 h. However, the role of HRT was evident at lower temperatures (5°C-10°C), when with the lower HRT, the COD removal efficiency also decreased (Bodik et al., 2002).

According to Young (1991), tests with laboratory and full-scale anaerobic filters that operated under a variety of conditions identified HRT as the most important design and performance parameter. In addition, influent waste strength and reactor height had essentially no effect on treatment efficiency when operating at a given HRT (Young, 1991). Taking these points into account, the following equation was proposed to correlate the total COD removal efficiency (E_t) with the *HRT*:

$$E_t = (1 - e^{-\epsilon/HRT}) \quad (1)$$

where ϵ is an empirical constant and E_t is the efficiency of the process calculated from the experimental total COD values.

However, assuming that not all the substrate is biodegradable, equation (1) can be transformed as follows:

$$E_t = a(1 - b/HRT^c) \quad (2)$$

where a , b and c are empirical constants obtained from the non-linear correlation of the experimental (E_t , HRT) pair values.

Therefore, by fitting the experimental data to equation (2), the following was obtained (Figure 2):

$$E_t = 0.8(1 - 2/HRT^2) \quad (3)$$

Given that the experimental method used to determine the COD does not distinguish between biodegradable and non-biodegradable COD_T, the non-biodegradable COD_T (COD_{T non-biod.}) was estimated by means of the extrapolation of effluent COD_T (S_e), which corresponds to an infinite HRT (Borja et al., 2002). Figure 2 illustrates this determination, the concentration of the COD_{T non-biod.} being equal to 117 mg/L.

Figure 3 plots the experimental biodegradable COD removal efficiencies (E_b) as a function of the HRT . As can be observed, for HRT s higher than 10 h, the COD removal efficiency obtained from the biodegradable COD (E_b) was 100%.

According to the literature to correlate the efficiency and HRT values it was assumed that degradation of substrate fits a first-order steady-state kinetic model (Kobayashi et al., 1983; Young, 1991). However, additionally, hydraulic factors such as the type of flow inside of the reactor (plug-flow mode or continuous stirred tank reactor, CSTR) must be known. Early tests with anaerobic filters suggested that upflow anaerobic filters operated in a plug-flow mode, and tracer tests in clean, media-filled reactors did show a pattern characteristic of plug-flow systems (Young, 1991).

However, other tests have shown the presence of considerable mixing, and intensive studies by Young and Young (1988) showed that even small amounts of gas flow definitely cause the liquid phase of the reactor to be essentially completely mixed, especially above the concentrated sludge zone. Data from full-scale reactors support this observation (Young and Yang, 1989). As a consequence of these two criteria, both the plug-flow and CSTR models were assayed to correlate the efficiency (obtained from the biodegradable COD) with the HRT (Scott Fogler, 1999):

$$\text{Plug-flow model: } E_{b_{pf}} = 1 - \exp(-k_{pf}HRT) \quad (4)$$

$$\text{CSTR model: } E_{b_{CSTR}} = 1 - 1/(1 + k_{CSTR}HRT) \quad (5)$$

where $E_{b_{pf}}$ and $E_{b_{CSTR}}$ are the COD removal efficiencies obtained from the biodegradable COD for the plug-flow and CSTR models respectively and k_{pf} and k_{CSTR} are the respective kinetic coefficients.

Figure 3 shows the plots of the fit of the experimental data (E_b , HRT) to the two proposed models (plug-flow and CSTR). The same Figure also illustrates the fit of the experimental data to the Young and McCarty model, which is given by equation (2) (Kobayashi et al., 1983). It can be inferred from Figure 3 that the flow pattern observed in the reactor studied is half-way between plug-flow and CSTR systems, although the plug-flow is predominant. On the other hand, it can also be observed that the Young and McCarty model almost coincides with that CSTR model, when the biodegradable COD is used for fitting the data.

3.3. Correlation of the Efficiency with the operation variables

While pilot-scale and laboratory-scale tests can be used to determine specific relationships between various design and operational factors – for example, reactor configuration, media type and placement, organic loading and HRT – no comprehensive

performance relationship that is accepted widely for design of full-scale anaerobic filters has been developed. Two different equations, among others, have been proposed in the literature to relate the COD removal efficiency with the operation variables:

According to Young (1991):

$$E = k (HRT)^a (S_0)^b (A_s)^c \quad (6)$$

where E is the COD removal efficiency (based on total COD), S_0 is the influent substrate concentration, A_s is the specific surface area, and a , b and c are constants characteristic of the assayed system. The slight effects of influent substrate concentration within the S_0 range studied may be attributable to the complete mixed nature of the liquid phase within anaerobic filters as was indicated previously (Young, 1991).

According to Manariotis and Grigoropoulos (2008):

$$S_e = a' S_0^{b'} (HRT)^{c'} P^{d'} \quad (7)$$

where S_e is the effluent substrate concentration, P is the porosity of the packing media and a' , b' , c' and d' are typical constants of the assayed system.

In the present work and given that the experimental phase was carried out with only one type of packing medium, both the variable A_s of equation (6) and P of equation (7) can be included in the corresponding constants of both models.

On the other hand, and taking into account that the efficiency, E , can be defined according to the following equation:

$$E = (S_0 - S_e)/S_0 \quad (8)$$

The equations of Manariotis can be expressed as follows:

$$E = 1 - K S_0^\alpha (HRT)^\beta \quad (9)$$

where α and β are constants characteristic of the type of packing media and wastewater used.

In the same way, the Young equation can be transformed into the following:

$$E = K' (HRT)^n S_0^q \quad (10)$$

where n and q are constants characteristic of the type of support material and wastewater used.

Figures 4 and 5 show the fit of the experimental data to equations of Young (eqn 10) and Manariotis (eqn 9) respectively, as well as the constants obtained in both cases. As can be observed the Manariotis equation allowed a better regression coefficient to be obtained showing a good fit of the experimental points. Therefore, the use of the Manariotis equation to estimate reactor performance would be a valuable tool in designing the anaerobic filter operation.

Finally, and taking into account that the effect of the initial COD concentration on the process efficiency is basically negligible, the equations (9) and (10) could also be further simplified by dropping the term S_0 and transformed into the equations (11) and (12) respectively, as follows:

$$E = 1 - K (HRT)^\beta \quad (11)$$

$$E = K' (HRT)^n \quad (12)$$

where the values of K and K' were found to be 0.4706 and 0.6156 h⁻¹, respectively. In addition, the constant β , included in the modified Manariotis model, reached the value of 0.3325 which is quite similar to the value previously obtained (0.3233). Simultaneously n achieved the value of 0.1010 in the modified Young model and 0.096 in the non-modified model. The similarity between the exponential values in both models shows the negligible effect of the initial COD concentration within the HRT range studied.

3.4. Methane yield coefficient

Figure 6 shows a plot of the daily methane production against the amount of COD consumed. By fitting the (methane production, g COD_T removed) value pairs to a straight line the average yield coefficient under standard temperature and pressure conditions (STP: 1 atm and 0°C) was found to be 0.15 L CH₄ STP/g COD_{removed}. As can be seen in Figure 6, the intercept of the linear plot is slightly negative. This suggests that when a small part of substrate is consumed the produced methane is not quantified. Some authors (Kobayashi et al., 1983) have suggested that energy recovery potential for filters operating at low loading rates is further diminished due to soluble methane loss in the filter effluent. On the other hand, it is also possible that a dragging of small bubbles of methane can also be produced. Lower methane yield coefficients (0.055 L CH₄ STP/g COD) were achieved in anaerobic filters treating low strength domestic wastewater (COD: 288 mg/L) at 24 h HRT and 25°C and 35°C. In contrast, higher methane yield coefficients (0.25 L CH₄ STP/g COD) were obtained in UASB reactors treating low strength synthetic wastewater (COD: 1000 mg/L) operating at a low temperature (15°C) and a HRT of 24 h (Akila and Chandra, 2007) when an inoculum from a cattle manure digester adapted to 15°C was used. Therefore, it appears that the microbial consortia acclimatized at a low temperature, used by most of the researchers for biomethanation at psychrophilic are not true psychrophiles. This has been inferred by the fact that true psychrophiles will not survive at an increased temperature. Most of the studies indicate an increase in gas production with the increase in temperature. Thus, it can be deduced that these are psychrotrophs (organisms that can withstand thermal fluctuations). Some of the common characteristics shown by these microorganisms during the acclimatization process decrease in the number of ion pairs, the side chain

contribution to the exposed surface and the apoplar fractions of the exposed surface
(Kashyap et al., 2003).

4. CONCLUSIONS

The psychrophilic (15°C-17°C) anaerobic digestion of synthetic domestic sewage
in an upflow filter was very stable for HRTs in the range of 3.6 h-17.1 h. For HRTs of
between 10.0 and 17.1 h, the total COD removal efficiency was virtually independent
on the HRT, achieving an average value of around 80%.

The flow pattern observed in the reactor was intermediate between plug-flow and
CSTR systems. The Young and McCarty model almost coincided with the CSTR
model, when the biodegradable COD was used. The Manariotis equation allowed a
better fit of the experimental data than the Young model.

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Table 1

Composition of the synthetic domestic wastewater.

Macronutrient Solution			Micronutrient Solution	
Compound	Source	[mg/L]	Compound	[mg/L]
Starch	C-Carbohydrate; 80%	200	FeCl ₃ ·4H ₂ O	1000
Ovoalbumine	C-Protein; 10%	21.0	CoCl ₂ ·6H ₂ O	1000
Sunflower oil	C-Lipid; 10%	13.1*	MnCl ₂ ·4H ₂ O	250
Urea	N	13.0	CuCl ₂ ·2H ₂ O	15
KH ₂ PO ₄	P	5.26	ZnCl ₂	25
CaCl ₂ ·2H ₂ O	Ca	22.05	H ₃ BO ₃	25
MgSO ₄ ·7H ₂ O	Mg	0.43	(NH ₄) ₆ Mo ₇ O ₂₄ ·4H ₂ O	45
KCl	K	21.3	NaSeO ₃ ·H ₂ O	50
NaHCO ₃	Na	8.76	NiCl ₂ ·6H ₂ O	35
Yeast Extract		100	EDTA	500
Micronutrients		1.0*	HCl 36 %	1*
			Resarzurin	250

* Amount expressed in mL/L.

Table 2

Characteristics of the synthetic domestic wastewater used and of the real domestic wastewater reported in the literature (Kobayashi et al., 1983; Elmitwalli et al., 2002).*

Parameter	Synthetic domestic wastewater used**	Real domestic wastewater (range of values)
Total COD (COD _T)	705 (30)	180 - 1100
Total Suspended Solids (TSS)	560 (22)	160 - 625
Volatile Suspended Solids (VSS)	530 (20)	80 - 580
pH	6.8 (0.3)	5.7 - 8.9
Volatile Fatty Acids (VFA) (as acetic acid)	130 (5)	90 – 150
Total Alkalinity	315 (15)	190 – 360

* All amounts, except pH, are expressed in mg/L.

** Values in brackets correspond to the standard deviations of the mean values.

Table 3

Values of the effluent pH and VFA/Alkalinity ratio for the different HRTs assayed.

HRT (h)	pH	AV/Alkalinity (equiv. acetic acid/equiv. CaCO ₃)
3.20	6.95	0.51
3.60	7.10	0.41
4.10	7.20	0.38
5.00	7.39	0.39
5.50	7.38	0.29
7.00	7.30	0.39
7.50	7.39	0.28
8.70	7.29	0.29
10.00	7.31	0.25
10.50	7.35	0.30
12.00	7.30	0.27
12.50	7.32	0.26
13.50	7.35	0.20
14.00	7.40	0.19
15.00	7.42	0.18
17.10	7.39	0.20

FIGURE CAPTIONS

Figure 1. Variation of the total COD removal efficiency (per unit) as a function of the HRT.

Figure 2. Estimation of the fraction of non-biodegradable organic matter (total COD) contained in the synthetic domestic wastewater used in this study.

Figure 3. Variation of the experimental efficiency values (based on biodegradable COD) with the HRT. The Figure also plots the theoretical plug-flow, CSTR and Young and McCarty models.

Figure 4. Variation of the COD removal efficiency with HRT and influent substrate concentration (Young model, eqn. 10).

Figure 5. Variation of the COD removal efficiency with HRT and influent substrate concentration (Manariotis model, eqn. 9).

Figure 6. Determination of the methane yield coefficient.

$$E = 0,8 \left[1 - \frac{1,99}{HRT^{2,02}} \right]$$

Figure 1

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603 **Figure 2**

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616 **Figure 3**

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$$E = 0.15 \cdot (\text{HRT})^{0.096} \cdot (S_0)^{0.2167}$$

627 $R = 0.87$; Standard Error of Estimate = 0.025

628 **Figure 4**

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$$E = 1 - 5.18 \cdot (\text{HRT})^{-0.3233} \cdot (S_0)^{-0.3686}$$

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R = 0.90; Standard Error of Estimate = 0.022

639 **Figure 5**

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methane yield coefficient

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647 **Figure 6**

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