ELSEVIER

Bioresource Technology 69 (1999) 103–115

BIORESOURCE TECHNOLOGY

Degradation of unsorted municipal solid waste by a leach-bed process

S. Chugh[a,b], D.P. Chynoweth[c], W. Clarke[a,b], P. Pullammanappallil[a,b], V. Rudolph[b]*

[a]*CRC for Waste Management and Pollution Control Ltd, Department of Chemical Engineering, The University of Queensland, St Lucia, QLD 4072, Australia*

[b]*Department of Chemical Engineering, The University of Queensland, St Lucia, QLD 4072, Australia*

[c]*Agricultural Engineering Department, Rogers Hall, University of Florida, Gainesville, FL 32611, USA*

Received 2 July 1998; revised 14 October 1998; accepted 20 October 1998

Abstract

In current landfills breakdown of municipal solid waste (MSW) occurs slowly and the landfill leaves a legacy of care, management, monitoring and potential catastrophic failure over several generations. Social concern over these long term issues, with their legislative and economic implementation, increasingly favour practices which promote short stabilisation times and minimise environmental impact. This paper describes experiments carried out on mixed and unsorted municipal solid waste (MSW) in which 75% of the rapidly biodegradable fraction was degraded in about 2 months with an average yield of 0.18 m^3 CH_4/kg volatile solids at s.t.p. The experiments served to demonstrate that with proper leachate management very rapid decomposition of waste can be accomplished by taking the waste through a series of controlled degradation stages.

Keywords: Anaerobic digestion; Municipal solid waste (MSW); Methanogenesis; Bioreactor; Landfill

1. Introduction

Sanitary landfills represent a common, economical and environmentally acceptable method for the disposal of solid wastes. Even with implementation of waste reduction, recycling and transformation technologies, the disposal of residual solid waste in landfills still remains an unavoidable component of an integrated solid waste management strategy. However, due to the unpredictable nature of the processes involved during the stabilisation of the waste in a landfill and differences in the waste composition, identification of the key parameters controlling waste degradation in a landfill is difficult.

There are several concerns with the current landfilling process relating to its potential for air and water pollution. Fugitive release of landfill gases occurs even in highly engineered systems with active gas extraction. Besides fire and explosion hazards, odour and greenhouse gas emissions remain difficult issues at most sites. Methane, which is a major product of biological processes taking place in the landfills, is recognised as a significant greenhouse gas (Rodhe, 1990). Landfill sites also pose another major threat to the environment, namely the potential loss of leachate to the surrounding water and soil. The chemical compositions of the leachates vary, depending on the age of landfill and the quality of waste disposed of in the particular landfill. The variability is further exacerbated by the fact that leachate being generated at any point in time is a mixture of leachates derived from solid wastes of different ages. The leachate may carry toxic contaminants to underground water supplies (Christensen et al., 1994). Unassisted, natural degradation in landfills occurs very slowly, and may continue over scores of years (Belevi and Baccini, 1992). Since the landfills present potential environmental threats over this entire period, care, monitoring and management of the sites are required for long periods, even after the sites are closed.

*Corresponding author. Tel.: +61-7-3365-3616; fax: +61-7-3365-4199; e-mail: victorr@cheque.uq.edu.au

0960-8524/99/$ — see front matter
PII: S0960-8524(98)00182-5

A number of factors contribute to the slow rate of waste degradation, including moisture limitation, poor shredding of waste, high bulk density and lack of inoculum. These have been discussed by Barlaz et al. (1990).

The average moisture content of freshly placed refuse is typically between 20 and 40%, compared with waste field capacity values of about 60%. A prolonged adjustment period in the landfill therefore occurs while moisture accumulates. The only sources of moisture in a conventional landfill are precipitation and the water that may be produced chemically during the waste decomposition process. Past investigations have shown that the addition of water to raise moisture content to field capacity accelerates waste stabilisation processes and stimulates early production of methane (Leckie et al., 1979; Wujcik and Jewell, 1980; Farquhar and Rovers, 1973; Rovers and Farquhar, 1973). Therefore, encapsulation of waste, in order to prevent ingress of moisture, which is a common practice in modern landfills, serves to retard the waste stabilisation process.

The complete degradation of the organic fraction of MSW under anaerobic conditions requires the concerted action of several groups of microorganisms. Degradation in a landfill will be delayed if the microorganism populations are not balanced. The methane-forming microorganisms grow at a rate that is much slower than the acid formers. This, together with the fact that methane-forming microorganisms cannot directly consume landfill waste, means that the acid formers will normally outgrow the methane formers. As a consequence, the degradable fraction of landfilled waste will normally become acidic, which slows down microbial activity and inhibits further degradation. Studies carried out by Buivid et al. (1981), Stegmann (1983) and Chynoweth et al. (1992) report that inoculation helps the onset of methanogenesis, by providing a balanced community of microorganisms.

The objective of this work was to achieve accelerated degradation of unsorted MSW by applying techniques that may be viable in a full-scale landfill. The experimental procedures and results discussed here were carried out to investigate a process to accelerate municipal solid waste degradation, where leachate is exchanged between a batch of existing anaerobically-degraded waste and a batch of fresh-waste. Such a process was first used by Chynoweth et al. (1992) to enhance degradation of sorted MSW under thermophilic conditions. The process arrangement serves three purposes. Firstly, the volatile fatty acids (VFAs) produced by the fresh-waste (which reduce the system pH) are flushed out into the leachate; the acids are then removed. Secondly, a stabilised-waste reactor provides a convenient site for the consumption of high-strength leachate generated by the fresh-waste reactor. Thirdly, the leachate, when passed through the stabilised-waste reactor, carries the inoculum to seed the fresh-waste to speed up the degradation.

2. Methods

2.1. Experimental set-up

Experiments were carried out in insulated, 200 l, 316 stainless steel reactors. Each reactor was provided with a built-in leachate collection tank, with a holding capacity of 42 l. The reactors were subjected to operating conditions approximating to those of large-scale landfills. These conditions included mesophilic temperature, and the use of unsorted and well representative raw waste feedstock, as received by a transfer station. Average packing density of waste in the reactors was about 500 kg/m^3. Figure 1 shows a schematic diagram of the reactor set-up.

Labtech Pro software (a product of Laboratory Technologies Corporation, USA) was used for on-line monitoring and control of digesters. The temperature of waste was maintained at 38°C, controlled by using a heating tape (a 450 W, KTeS series type, manufactured by ISOPAD GmbH, Heidelberg, Germany). Each reactor, including the leachate collection tank was heated as a single unit. Each reactor was equipped with three thermocouples, measuring temperatures at three separate radial and vertical positions in the waste. A separate thermocouple was used for monitoring temperature of leachate in the collection tank. All thermocouples recorded similar readings.

A proportional–integral control algorithm was used to control the temperature of the reactors, with the temperature of the waste near the wall of the reactor as the controlled variable. The control algorithm was tuned so that temperatures were maintained within 2°C of the setpoint. A ramp and soak technique was used to heat a reactor over several hours from the initial room conditions. This was done to avoid any sudden temperature shock to the microorganisms in the waste bed and overheating of waste near the surface of the reactors.

The biogas production from each reactor was measured using a displacement gasmeter. The gasmeter consisted of a U-tube, made from perspex (acrylic), a relay, a float switch, a timer, a debounce module (to overcome false readings due to chattering of the relay), a counter and a solenoid valve. Silicone Fluid (200 Fluid, 50 centistokes, manufactured by Dow Corning) was used as the fluid for the U-tube due to its low vapour pressure and low solubility towards landfill gas components. The biogas from the reactor accumulated in one limb of the U-tube and displaced the liquid in it. When the liquid in the second limb rose to

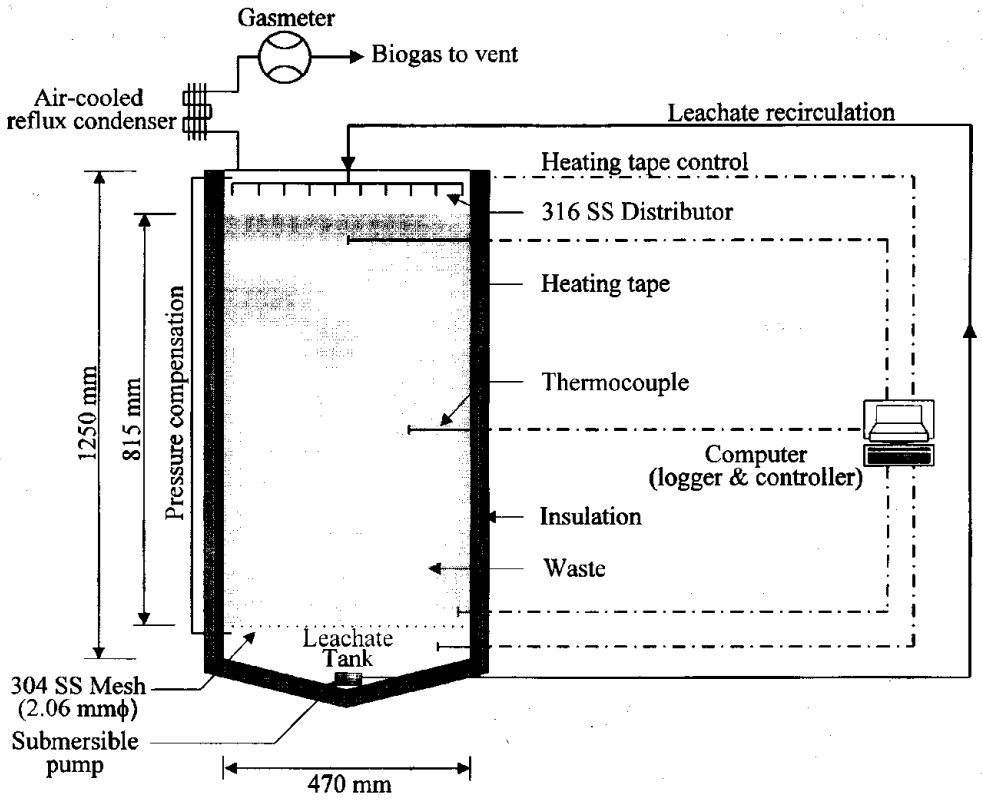

Fig. 1. Schematic diagram ofreactor set-up.

a certain level, the float switch tripped which caused three events to happen simultaneously — a signal was sent to a counter to record the reading for display; the biogas from the first limb was vented to atmosphere (via a biogas collection manifold) through the solenoid valve to reset the liquid levels in both limbs; and the timer was activated to keep the vent line on the solenoid valve open to allow the accumulated biogas to escape. The duration for which the timer kept the vent line open was manually set to vent out the accumulated biogas in the first limb. The 3-way solenoid valve isolated the reactor from the gasmeter during the vent cycle. A simple air cooled stainless steel coil condenser was installed at the biogas outlet of the reactor, before the gasmeter, to trap condensate and reflux it back to the reactor vessel. This was necessary to prevent moisture collecting in the gasmeter and to obtain moisture-free gas samples for Gas Chromatograph analysis.

Leachate pumping was carried out using polypropylene submersible pumps (Tauchpumpe, Nr. 511.0412, supplied by Mocar GmbH, Hamburg, Germany). The amount of leachate to be recirculated daily was fixed as a predetermined percentage of the initial volume of waste loaded in the reactor for each run. This leachate volume was recirculated in one batch, at a flowrate of about 2.2 l/min. Pumping the leachate into the reactors took about 7 min. This leachate then percolated through the waste bed, and accumulated in the leachate tank, over the next 24 h.

2.2. *Analytical procedures*

Total solids analysis was measured (APHA, 1992) by drying the sample in an oven at 105 ± 1°C. Before any further analyses were carried out, the dried waste was finely shredded in a cross-beater mill to an average particle size of about 2 mm, in order to obtain a homogenous sample. A riffle splitter was then used to mix the waste thoroughly and reduce the sample size to the desired amount of about 750 g (dry mass). Multiple subsamples were then randomly taken from this sample for each subsequent analysis that was carried out on the solids. Volatile solids were measured using APHA (1992) by ashing the dried waste in a furnace at 560 ± 5°C.

Gas composition analysis was done on a Perkin Elmer Autosystem Gas Chromatograph, fitted with a Porapak Q 80/100 mesh column and a 10 μl sampling loop. Nitrogen was used as a carrier gas. Gas samples were eluted into a thermal conductivity detector, and analysed for hydrogen, methane and carbon dioxide.

All inorganic analyses on leachate were out carried using the spectroquant analysis system, on a Merck photometer SQ 118. For digestion and heating of samples, a Merck thermoreactor TR 300 was used.

Volatile fatty acids (VFAs) were analysed using a Perkin Elmer Autosystem Gas Chromatograph, fitted with a J&W Megabore column, DB-FFAP with a length of 30 m, internal diameter of 0.53 mm and film thickness of 1 micron. Nitrogen was used as a carrier gas. The leachate samples were eluted into a flame ionisation detector, and analysed for acetic, propionic, iso-butyric, butyric, iso-valeric, valeric and hexanoic acids.

The Biochemical Methane Potential (BMP) technique, developed by Owen et al. (1979) and later adapted by Owens and Chynoweth (1993) for MSW, was used to determine sample biodegradability and the extent of the decomposition process during the operation of the reactors. This technique distinguishes between the volatile solids contributed by the biodegradable fraction and the volatile solids contributed by the non-biodegradable fraction of a sample.

2.3. Experimental procedures

2.3.1. Feedstock preparation

In order to overcome the inherent, stochastic heterogeneity in collecting waste in different batches for each experiment, a single sample of about 2 t was collected from a local transfer station, from a much larger bulk that had been thoroughly mixed by a front-end loader. The waste was shredded in an industrial shredder to an average particle size of about 10 cm. In order to avoid degradation of the collected waste at ambient temperatures, it was then loosely packed in 120 l black polypropylene drums with gas-tight lids and stored in a local industrial freezer at a temperature of about −28°C. The drums were removed approximately 14 h prior to the loading of reactors, to allow sufficient time for thawing of the waste. An average of two drums was used to load a reactor.

2.3.2. Solid sample preparation

The solid samples were obtained only at the beginning and at the end of each experiment. To characterise the waste, a sample of at least 1.5 kg (dry mass) was collected. The moisture content of the waste from the previous studies was used for the sample size calculations. The same sampling protocol was used for both the waste that was loaded into the reactors and the degraded waste unloaded from the reactors. Since more than one reactor was normally loaded at a time, a common sample was taken for analysis. At the time of unloading, the waste from each reactor was sampled separately for analysis.

2.3.3. Operation of reactors

The process investigated here involved two-stage leachate recycle (Fig. 2). The initial leachate generated from fresh-waste (Reactor B) was fed through a waste that had already been stabilised (stabilised-waste reactor, Reactor A). Stabilised-waste is defined here as the waste that had been taken through its various stages of anaerobic degradation and exhausted of its methane-producing potential. The leachate from fresh-waste was low in buffering capacity and high in chemical oxygen demand (COD). The stabilised-waste contained a varied consortium of microorganisms, which converted the organic carbon in this leachate into methane and carbon dioxide. The leachate from the fresh-waste, having passed through the stabilised-waste, carried inoculum back to the fresh-waste. Once a balanced microbial community had been established in the fresh-waste, indicated by a pH value of 6.5 of leachate, the leachate from the fresh-waste was recirculated, without taking it through the stabilised-waste reactor and the degradation process in the fresh-waste reactor proceeded to completion. The pH was chosen as a control variable because it was easy to measure, and provided a meaningful and easily-interpreted signal. The decision to uncouple the fresh-waste and stabilised-waste reactors at a pH value of 6.5 was made on the basis of environmental factors favourable for methanogenic activity. Past studies had shown that methanogenesis was favoured at a pH between 6.4 and 7.2 (Farquhar and Rovers, 1973). Once the fresh-waste reactor was digested to completion, it was then used as a stabilised-waste reactor to start degradation in another batch of fresh-waste (Reactor C).

2.3.4. Start-up experiments

The study included a start-up phase, in which an anaerobic digester was started by seeding MSW with anaerobically digested sewage sludge (dry matter content of about 4%). This was followed by a series of experiments where only the effluent leachate from a stabilised-waste reactor was used to start degradation in a fresh-waste reactor, proceeding down through generations. Figure 3 shows the experimental plan carried out in order to investigate the process. The first reactor, seeded with the sludge, was designated as a 'zero'-generation reactor. The next reactor started up by using the leachate from this zero-generation reactor was a 1st generation reactor, and so on. Since the microbial community present in the sludge was different from the one required for the degradation of MSW, this series of experiments served to establish that the maximum degradation rates, based on a fully evolved and acclimatised microbial community, were achieved for the substrate available in MSW.

The degradation of the first batch of shredded waste was initiated by inoculating it with sludge from a mesophilic sewage sludge digester (Experiment 1). Alternate layers of sludge and waste were placed in the reactor, giving a sludge to waste ratio of 1:1 by weight. Distilled water was added to this reactor to produce

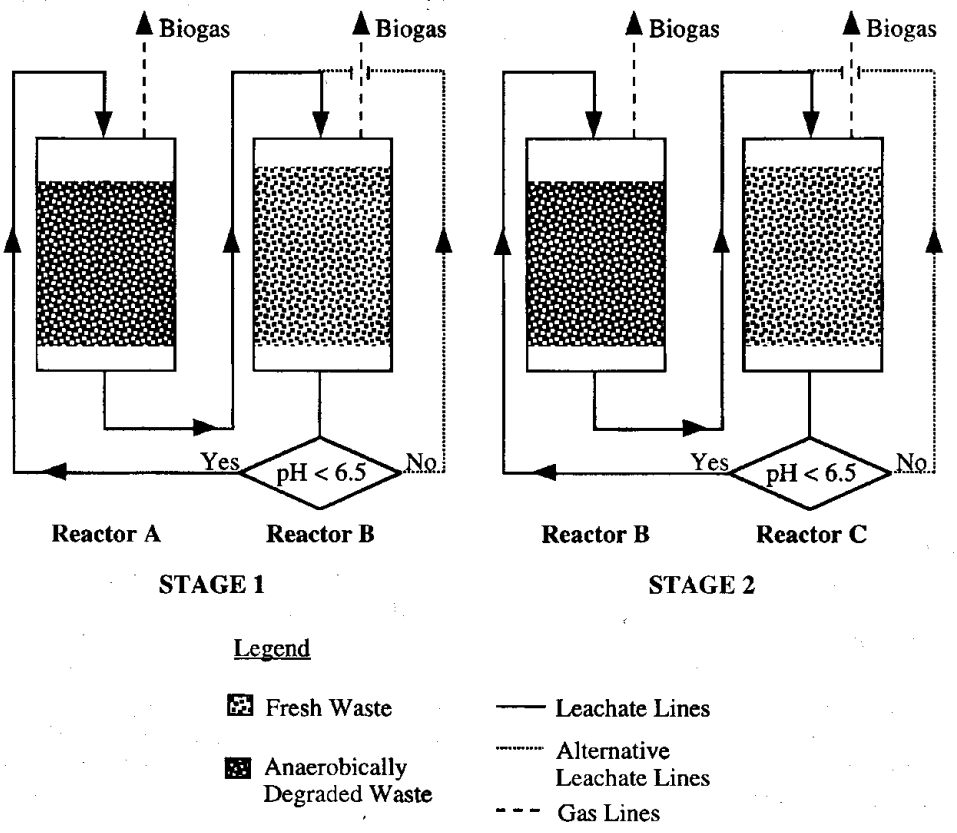

Fig. 2. Schematic diagram of the proposed process.

the set amount of leachate required for the recirculation. The leachate was recirculated back over the waste (called here 'direct recirculation') after adjusting its pH. The leachate recirculation was carried out until the waste in the reactor stabilised, i.e. biogas production had stopped after attaining a maximum and the pH of the effluent leachate had stabilised at around 7.0.

A second reactor (fresh-waste reactor, Experiment 2) was prepared using fresh coarsely shredded MSW only. The waste was slowly wetted using distilled water, to a point where liquid had drained through. This allowed the leachate recirculation to be commenced immediately. The temperature of the waste bed was then taken to 38°C and recirculation started. The exchange of leachate (called here 'indirect recirculation') between the stabilised-waste and the fresh-waste could not be started immediately. This was because the sludge reactor, which was to be used as the source of inoculum took some three weeks longer to achieve stability than was anticipated. The fresh-waste reactor was recirculated directly during this period. When the sludge reactor finally reached stability, indirect recirculation was carried out until the pH of effluent leachate from the fresh-waste reached a value ≥ 6.5, at which time direct recirculation commenced. This was continued until the waste and leachate were stabilised with respect to pH of the leachate and the biogas quantity.

In order to obtain two comparable first generation reactors, another fresh-waste reactor (Experiment 3) was started using the original sludge reactor as the stabilised-waste reactor, after it was no longer required for indirect recirculation in Experiment 2. The same leachate recirculation protocol was followed as in Experiment 2, with the exception that the indirect recirculation between the stabilised-waste and the fresh-waste commenced immediately after the fresh-waste reactor was prepared. The fresh-waste and stabilised-waste reactors were uncoupled when the pH of the effluent leachate from the fresh-waste reached a value ≥ 6.5, after which the fresh-waste reactor was directly recirculated to complete waste degradation. At the end of this experiment, two stabilised-waste reactors (from Experiments 2 and 3) had been established and made available for further work. This concluded the start-up phase. Since the sole purpose of the start-up phase was to achieve two reactors containing stabilised-waste, no detailed analyses were carried out.

2.3.5. *Subsequent experiments*

All the subsequent experiments were started by using the effluent leachate from one of these, or a later

generation of stabilised-waste reactor. The sludge reactor, having served its purpose in starting up two reactors containing MSW only, was then decommissioned.

In addition to the series of experiments discussed above, two further experiments were also carried out as control experiments. In the first experiment (Control Experiment 1), after loading the reactor with fresh-waste, the temperature of the waste bed was increased slowly to 38°C and then it was left alone, without the addition of moisture. This approximately represents current 'contained, dry landfill' practice. The second experiment (Control Experiment 2) was carried out in similar fashion to the other experiments in this study, with the exception that only the leachate from the fresh-waste reactor was directly recirculated, i.e. leachate from a stabilised-waste was not used to inoculate the fresh-waste.

3. Results and discussion

A total of six experiments was carried out to investigate the process (Fig. 3). This section provides details of one of the experiments to explain the salient features of the degradation process. Thereafter, each of the parameters which were monitored to indicate the rate and extent of degradation are discussed.

3.1. Detailed analysis

After the fresh-waste reactor was sealed, a sufficient quantity of distilled water was added to produce the set amount of leachate and the waste bed was slowly heated to 38°C. The experiment was started by coupling the fresh-waste reactor with an existing stabilised-waste reactor, obtained from a previous experiment. The indirect recirculation commenced immediately after the fresh-waste reactor was ready. Results from this experiment are shown in Fig. 4.

The pH of the effluent leachate from the fresh-waste rose from its initial value of 4.2, prior to the commencement of indirect recirculation, to 6.4 on day 2. Between day 2 and day 3, the pH dropped to 5.8. It then rose steadily to 6.8 by day 10 and then remained stable around that value for one day. From day 11 to 14, the pH rose steadily to about 7.2. From day 14 to day 29, the pH remained stable at a value around neutral. There was no significant change in the pH value of 7.4 between day 30 and 70.

The exchange of leachate between the fresh-waste and stabilised-waste removed VFAs from the fresh-

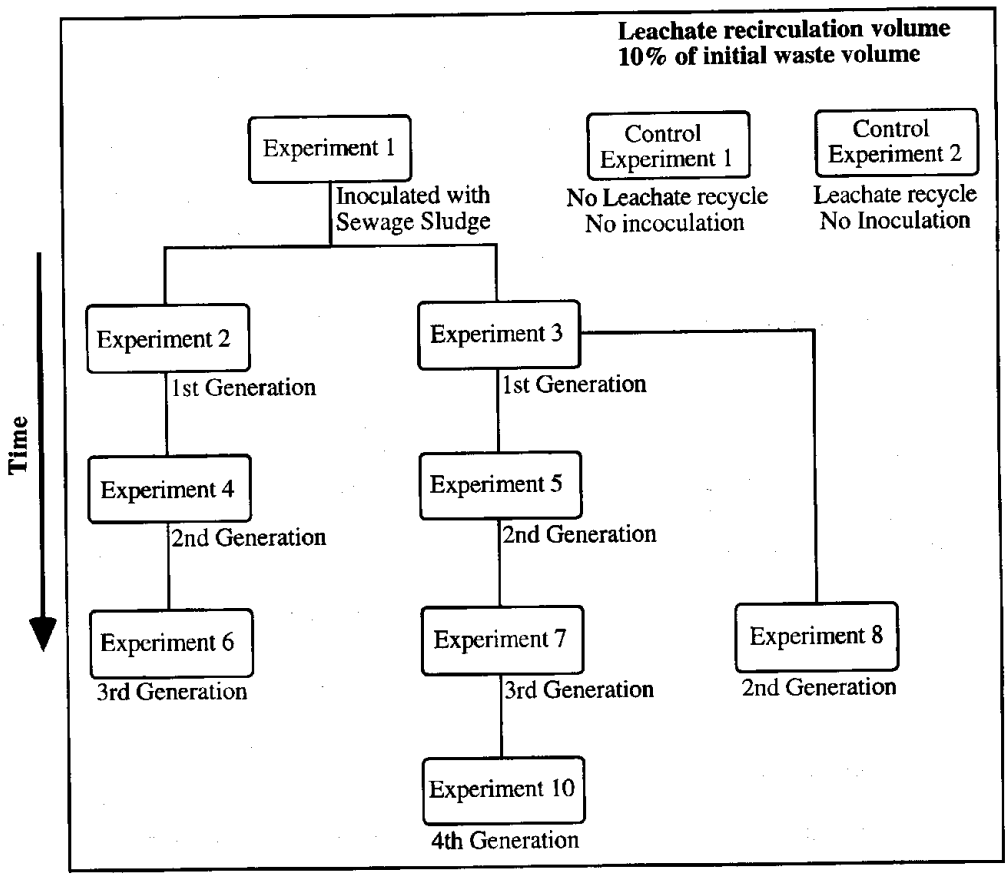

Fig. 3. Experimental plan to study the process.

waste through flushing. However, a higher pH in the fresh-waste accelerated acid production also, to such an extent that the pH value started to drop, due to accumulation of VFAs. Pohland and Kang (1974), and Robinson and Maris (1979) report that although control of pH and initial seeding enhance the decomposition of waste, these factors provide a favourable environment for the acid formers and are therefore unfavourable to the methane formers. However, the indirect recirculation continued to remove VFAs through flushing of the fresh-waste, and provided it with an inoculum. This is evident from the trends of leachate VFA concentration and biogas of fresh-waste in Fig. 4, which show that the VFAs from this waste bed fell with concomitant increase in biogas. McCarty (1964a–d) and WPCF (1987) report that VFAs and pH are related parameters that influence digester performance. Under conditions of overloading and in the presence of inhibitors, methanogenic activity cannot remove hydrogen and VFAs as fast as they are produced. At low pH, un-ionised species of VFAs are formed, consuming the bicarbonate alkalinity and carbon dioxide production increases, reinforcing the shift in VFAs towards the un-ionised state. Un-ionised

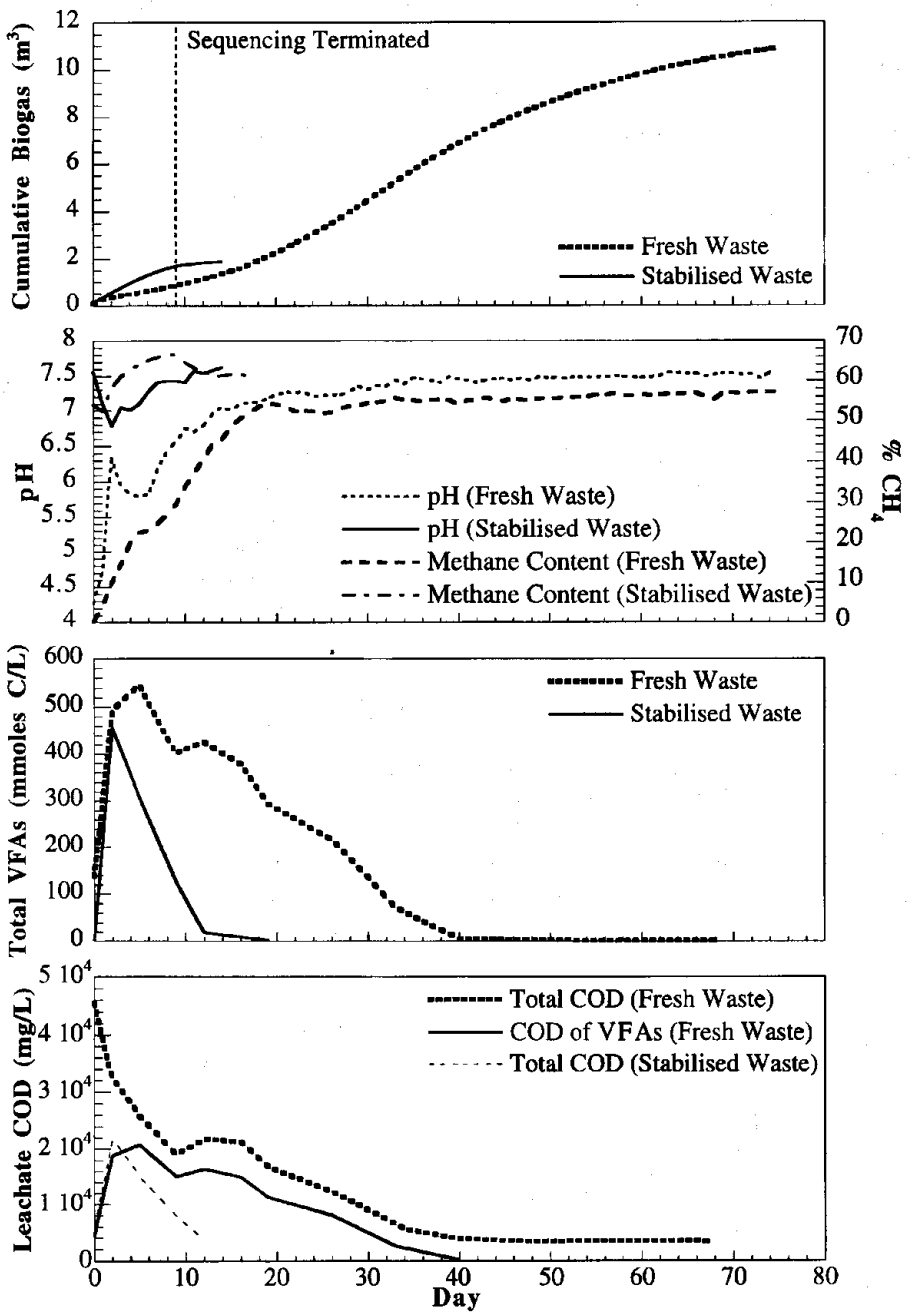

Fig. 4. Typical trends of monitored parameters.

species of VFAs are more toxic to methane formers than ionised species. McCarty (1964a–d) also reported that certain volatile fatty acids are associated with the onset of digester failure, including propionic and higher-molecular-weight fatty acids. WPCF (1987) reports that if the total acids cannot be changed, changing the pH and therefore changing the un-ionised concentration can be a useful way of preventing toxicity. The continued indirect recirculation between the stabilised-waste reactor and the fresh-waste reactor served both these purposes. The leachate from a stabilised-waste reactor flushes out the VFAs in a fresh-waste reactor, which are at toxic levels and also improves the buffering capacity by inoculating the fresh-waste bed. This results in removal of VFAs and an increase in the pH in the fresh-waste reactor. The two reactors were uncoupled after 9 days of indirect recirculation, when the pH reached a value of ≥ 6.5, in this case 6.6. The pH of the effluent leachate dropped initially but picked up, reaching neutral on day 12. The total VFAs also showed the same trend. The analysis of individual VFAs showed that the propionic and higher-molecular-weight fatty acids increased with the uncoupling of the reactors (Fig. 10). Since the pH value did not drop below a value of 6.5, these acids remained in the ionised state.

The cumulative biogas curves show that although the two reactors were coupled for 9 days, the biogas from the stabilised-waste stopped on day 8. This demonstrated that, at this stage, most of the COD conversion was taking place in the fresh-waste reactor. The COD drop was gradual until day 14. Once the pH reached neutral value, trends show rapid drops in total VFAs and COD. The uncoupling of the reactors also saw a rapid increase in biogas production. The methane content in the biogas from the fresh-waste reactor reached its final value of 56% on day 18.

The results indicate that though the uncoupling of the reactors caused some initial distress to the fresh-waste, its pH trend shows that the waste bed was sufficiently well inoculated and buffered to overcome this imbalance quite quickly. The trends of total COD and COD contributed by the VFAs show that there is always a certain amount of residual COD, which cannot be degraded. The trends also show that these two variables followed a remarkably close pattern and were almost parallel from day 6 when the total VFAs reached a maximum value. This clearly indicated that the residual COD was contributed by the indigestible substrate. Biochemical methane potential (BMP) assays were performed, using the leachate obtained from a stabilised-waste as a substrate. The results confirmed that no further degradation was achievable under the experimental conditions.

Figure 4 shows that the pH of the effluent leachate from the stabilised-waste remained stable at a value between 6.8 and 7.5, with methane content in the biogas from this reactor holding between 56 and 60%. This indicated that the microbial populations in the stabilised-waste were stable and robust.

The experiment was carried out for a total of 74 days, until the biogas dropped and there was no further reduction in total COD and total VFAs in the effluent leachate. A yield of 0.17 m^3 CH_4/kg VS at s.t.p and a VS reduction of 67.4% was obtained from this experiment.

3.2. Comparative analysis of key parameters

3.2.1. pH and biogas production

Figure 5 and Table 1 show that as the experiments progressed down through generations the start-up period decreased, as was indicated by reduction in the number of days required to uncouple fresh- and stabilised-waste reactors. However, the experiments carried out in generations 3 and 4 showed negligible reduction in this start-up period. The fresh-waste reactors in the 3rd generation experiments when compared with the experiments carried out in the 2nd generation took fewer days to reach a pH of 6.5, at which time the reactors were uncoupled, and also fewer to reach a neutral pH. This is further supported by the pH trend of Experiment 8, a 2nd generation experiment carried out about 2 months after Experi-

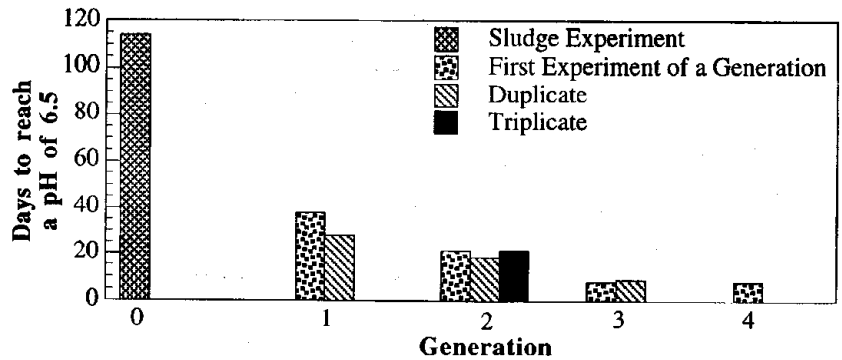

Fig. 5. Change in start-up period for degradation with generation.

Table 1
Process performances

Experiment	Number of days to uncouple reactors (days)	Volatile solids loaded in reactors (kg)	Total waste reduction (dry weight) (%)	Total VS reduction (dry weight) (%)	Methane yield $\left(\frac{m^3\ CH_4}{kg\ VS}\ \text{at s.t.p.}\right)$
4[a]	21	27.18±1.6	38.6±1.2	54.7±1.7	0.18±0.008
5[a]	18	27.66±1.9	37.0±1.4	54.8±1.3	(0.16–0.19)±0.009
6[b]	8	36.15±2.3	51.7±0.9	65.0±2.1	0.17±0.008
7[b]	9	37.11±1.9	57.1±1.6	67.4±2.3	0.17±0.007
8[a]	21	37.41±2.4	56.1±1.3	68.9±1.9	0.17±0.009
10[c]	8	29.8±1.8	42.3±1.2	57.6±1.1	0.18±0.01

[a]2nd generation experiment.
[b]3rd generation experiment.
[c]4th generation experiment.

ments 4 and 5, also 2nd generation experiments. The leachate pH from Experiment 8 followed a similar trend as in Experiments 4 and 5. Although an attempt was made to uncouple the fresh-waste and stabilised-waste reactors prematurely, Figure 6 shows that stable leachate pH could not be maintained and therefore, the fresh-waste reactor was recoupled with its stabilised-waste reactor. In addition, leachate pH in Experiment 10, a 4th generation experiment and carried out at another time, followed a trend similar to the Experiments 6 and 7, the 3rd generation experiments. These results demonstrate that reduction in start-up period depends on acclimatisation of micro-organisms, with the progress of the experiments.

Figure 7 shows the daily biogas production from the fresh-waste reactors for Experiments 4, 6, 7, 8 and 10. The cumulative biogas productions achieved from Experiments 6, 7, 8 and 10 are within 0.2 m^3, in a total yield of over 10 m^3. Due to a malfunction of the gasmeter, the daily biogas production from the fresh-waste reactor in Experiment 5 could not obtained. However, the results obtained from other analysis of this experiment show that all parameters being monitored followed similar trends as in Experiment 4. This indicated that the degradation of waste in Experiments 4 and 5 proceeded in a similar fashion.

The methane-yield curves show that the biogas production variation might have been due only to the variation in the waste composition (Fig. 8). Although the total biogas produced in Experiments 4 and 10 was less than the values obtained in Experiments 6, 7 and 8, once the data were normalised on the basis of the

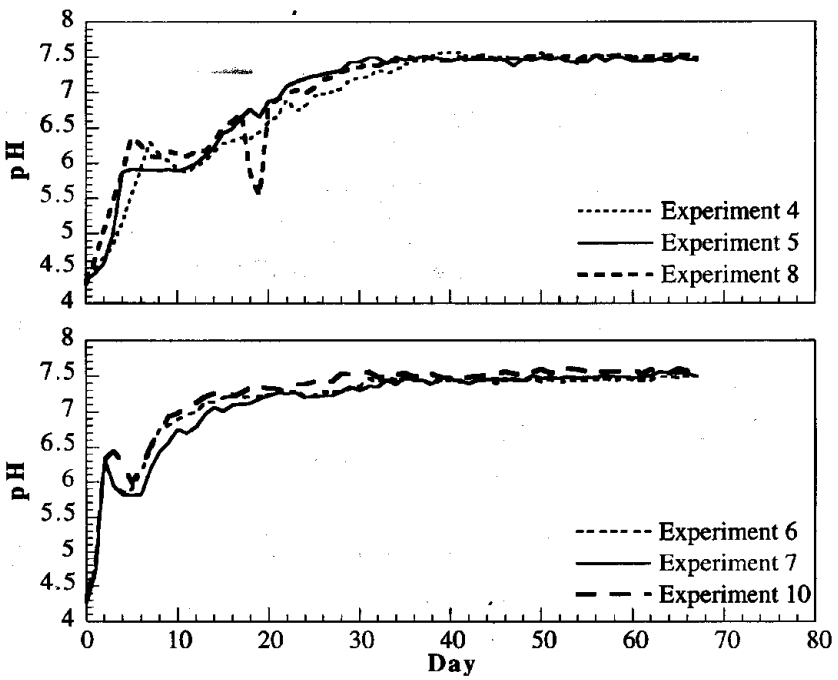

Fig. 6. pH of leachate from fresh-waste reactors.

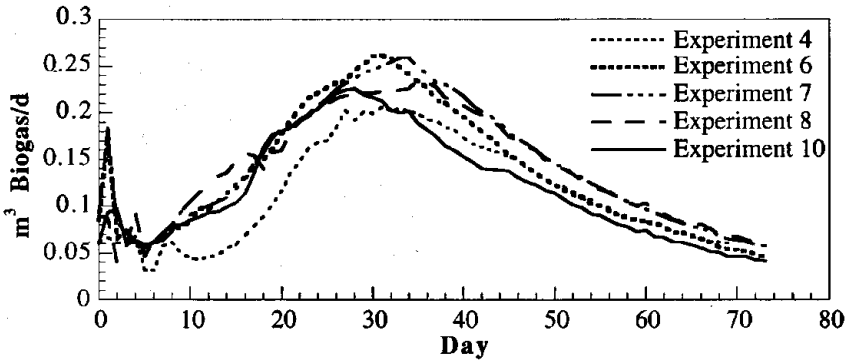

Fig. 7. Daily biogas production from fresh-waste reactors.

volatile solids added to each of the fresh-waste reactors, the curves were almost identical, indicating that these reactors behaved similarly. It should be noted that the drums of waste taken from the common lot in cold storage were mixed before being loaded into the reactors for concurrent experiments. Although waste taken at different times from the storage would have inevitably displayed some variability, waste loaded into reactors at the same time would be very similar. Thus, reactors for Experiments 4 and 5 were loaded from the same bulk batch; 6, 7 and 8 from another batch; and 10 from a third batch. The estimates of the biogas production from Experiment 5, obtained on the basis of the limited data, show that the yield was within the range of 0.16–0.19 m^3 CH_4/kg VS at s.t.p. This is similar to the results obtained from the other experiments.

At the commencement of indirect recirculation, daily biogas production from a fresh-waste reactor increased rapidly and then dropped. This trend was typical of all the experiments carried out here (Fig. 7). Figure 9 shows that this biogas consisted mainly of carbon dioxide, which is the major product of fermentation reactions. Christensen and Kjeldsen (1989) reported

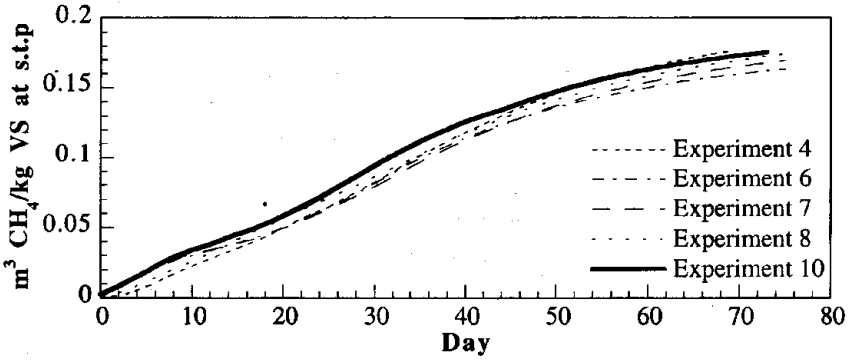

Fig. 8. Trends of cumulative methane yield.

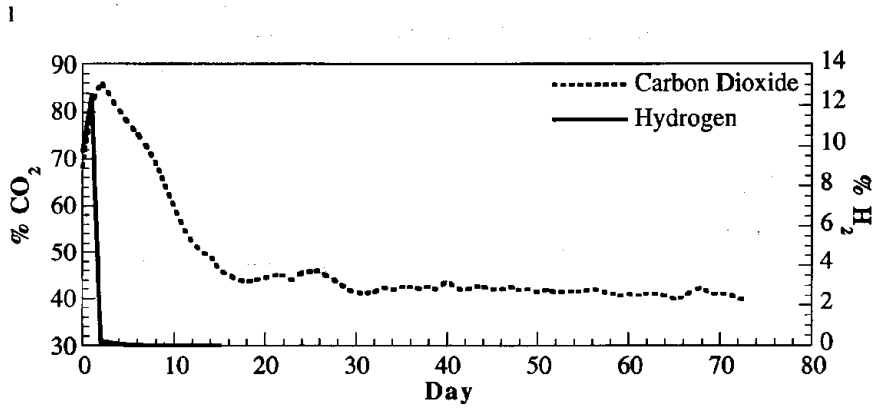

Fig. 9. Carbon dioxide and hydrogen trends.

that as the anaerobic stage develops, the activity of the fermentative and acetogenic microorganisms is high, producing high concentrations of carbon dioxide and hydrogen in the biogas, with high concentrations of volatile fatty acids in the leachate. The concentration of carbon dioxide reaches its peak value during the acid formation phase and can reach as high as 85%. In the current studies similar results were obtained, where carbon dioxide production was high during the peak production of volatile fatty acids, reaching values in the range of 75–85%. Hydrogen was also produced during the initial stages of anaerobic digestion, as shown in Fig. 9.

3.2.2. *Volatile fatty acids*

Figure 10 shows trends of individual volatile fatty acids for a typical run. At the start of an experiment, only acetic acid was detected. With the commencement of indirect recirculation, the seeding of fresh-waste started the hydrolysis and fermentation process. Rees (1980) reported that the leachate generated from freshly placed MSW contained mainly acetic acid. Due to the favourable environment for the acid formers, mainly high pH, other acids start to appear. These acids primarily consist of propionic, butyric, valeric and hexanoic acids, the products of digestion of carbohydrates. Butyric acid is a major acid formed by the hydrolysis of lipids. Concentrations of iso-butyric and iso-valeric acids were low. Iso-butyric and iso-valeric acids are primarily formed during the digestion of proteins. Rees (1980) explained that low quantities of these acids in leachate indicate that the protein content of the waste is also low. Propionic acid gave two distinct peaks in these experiments, with no accumulation in between. Kaplovsky (1951) obtained similar results during the digestion of fresh sewage solids. Kaplovsky (1951) explained that compounds such as lactic acid, produced during the digestion of solids, provide the source of the first peak and attributed the second peak to propionic acid production from more complex substances.

3.3. *Process performance*

The process showed consistent results for dry-solid weight reduction, volatile solids (VS) reduction and methane yield for all experiments. These results are summarised in Table 1. Volatile solids reduction was carried out after riffle splitting the sample and on average, 20 random subsamples were taken from this riffle-split sample. The values of volatile solids (VS) reduction for Experiments 4 and 5, and also 10, are lower than the values obtained for Experiments 6, 7 and 8. The explanation is that some batches of waste had less volatile solids than others, as shown by the consistent methane yield.

Biochemical Methane Potential (BMP) assays were carried out on the fresh-waste loaded into the reactors and the stabilised-waste removed from the reactors after the experiments were terminated. These results are summarised in Table 2. Although the results show that the maximum yield obtained for the loaded waste

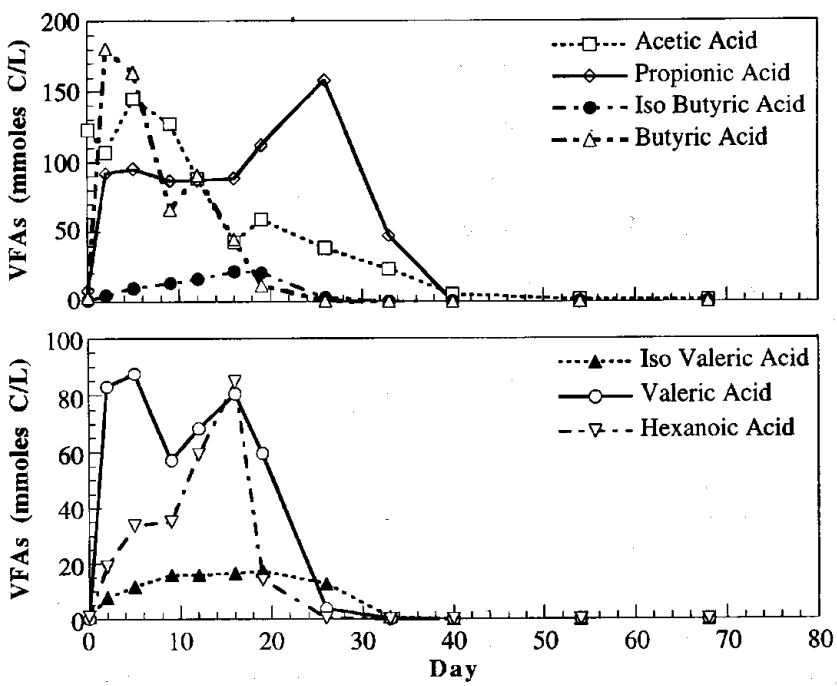

Fig. 10. Trends in individual volatile fatty acids.

Table 2
Comparison of methane yields from reactors and BMP assays

Experiment	Methane yield from feedstock (reactors) $\left(\frac{m^3\ CH_4}{kg\ VS}\ \text{at s.t.p.}\right)$	Methane yield from feedstock (BMP) $\left(\frac{m^3\ CH_4}{kg\ VS}\ \text{at s.t.p.}\right)$	Methane yield from residue (BMP) $\left(\frac{m^3\ CH_4}{kg\ VS}\ \text{at s.t.p.}\right)$
4	0.18 ± 0.008	0.23 ± 0.013	0.06 ± 0.003
6	0.17 ± 0.008	0.25 ± 0.015	0.07 ± 0.005
7	0.17 ± 0.007	0.22 ± 0.012	0.05 ± 0.003
10	0.18 ± 0.01	0.24 ± 0.016	0.08 ± 0.004

s.t.p. = standard temperature and pressure (273K and 1 atm).

varied from 0.22 to 0.25 m^3 CH_4/kg VS at s.t.p, these values were obtained after 108 days. After 60 days of incubation, the methane yield values obtained from these BMP runs were comparable to those achieved from the reactor study. The BMP runs on the stabilised-waste showed that the yield obtained from these samples varied between 0.05 and 0.07 m^3 CH_4/kg VS at s.t.p. Similar to the fresh-waste assays, the BMP runs on the unloaded waste were also carried out for 108 days; these assays produced little or no biogas during the first 60 days. The difference in the maximum methane yields for the BMP analyses and for the reactor studies might have been due to the presence of some refractory matter in the waste feedstock. The BMP assays incorporated favourable environmental conditions for the microorganisms, by addition of nutrients, buffer, inoculum and moisture. In addition, the temperature was controlled at a mesophilic level and the surface area of the waste was increased by shredding the substrate to an average particle size of 2 mm. Therefore, the BMP assays were used to determine the maximum amount of material in a sample that could be degraded by anaerobic organisms and represent a guide to the (best possible) target result, not necessarily achievable in a reasonable timeframe in practical systems.

At the completion of this study, 2 y after its commencement, the two control experiments remained incomplete. The effluent leachate from the Control Experiment 1 had a pH of 6.4 and a COD of 6700 mg/l, with no methane detected in the biogas. However, in the case of the Control Experiment 2, the leachate had a pH of 5.1 and a COD of 37000 mg/l. Methane content in the biogas, the total production of which had been very small, was 1.5%. The results from these experiments demonstrated that the degradation did not either commence (Control Experiment 1) or complete due to the failure of digester (Control Experiment 2), primarily due to the lack of sufficient moisture or inoculum limitation.

4. Conclusions

As the experiments progressed through the generations, the start-up period for degradation in a batch of fresh-waste became shorter. These experiments were continued until the degradations of fresh-waste followed a similar pattern. Although all experiments were performed on MSW that was collected from one 2 t grab, some variation in waste composition between loadings was apparent and impossible to eliminate completely. The digesters filled with MSW at the same time, from a single batch of mixed waste showed similar results, but the results varied when the digesters were filled at another time, with MSW from a different batch of mixed waste. However, once the data were normalised on the basis of volatile solids (VS), the results were similar for all loadings.

The process investigated here overcomes the disadvantages of a batch reactor by successfully starting a digester by inoculation with leachate. Once conditions are achieved, where the microorganisms are acclimatised to the environment in a fresh-waste bed, the start-up period is dramatically reduced to just a few days. The traditional biogas production curves which show erratic biogas generation from landfilled MSW become more uniform and steady. Accelerated degradation also implies enhanced biogas generation, under controlled conditions, which makes utilisation of the biogas generated from biodegradation of MSW economically more attractive.

The experiments reported here confirm that repeatable results can be achieved and the proposed process can successfully start a digester loaded with unsorted raw waste and successfully degrade it quickly.

Acknowledgements

This work was funded by the Cooperative Research Centre for Waste Management and Pollution Control

Limited, a centre established and supported under the Australian Government's Cooperative Research Centres Program. The authors also acknowledge Dr Andre Krol for his substantial contribution to this research program.

References

APHA, 1992. Standard Methods for the Examination of Water and Wastewater. American Public Health Association, Washington.

Barlaz, M.A., Ham, R.K., Schaefer, D.M., 1990. Methane production from municipal refuse: a review of enhancement techniques and microbial dynamics. Critical Reviews in Environmental Control 19, 557–584.

Belevi, H., Baccini, P., 1992. Long-term leachate emissions from municipal solid waste landfills. In: Cossu, R., Stegmann, R. (Eds), Landfilling of Waste: Leachate. Elsevier Applied Science, London, pp. 431–440.

Buivid, M.G., Wise, D.L., Blanchet, M.J., Remedios, E.C., Jenkins, B.M., Boyd, W.F., Pacey, J.G., 1981. Fuel gas enhancement by controlled landfilling of municipal solid waste. Resource Recovery and Conservation 6, 3–20.

Christensen, T.H., Kjeldsen, P., 1989. Basic biochemical processes in landfills. In Christensen, T.H., Cossu, R., Stegmann, R. (Eds), Sanitary Landfilling: Process, Technology and Environmental Impact. Academic Press, London, pp. 29–49.

Christensen, T.H., Kjeldsen, P., Albrechtsen, H.-J., Heron, G., Nielsen, P.H., Bjerg, P.L., Holm, P.E., 1994. Attenuation of landfill leachate pollutants in aquifers. Critical Reviews in Environmental Science and Technology 24, 119–202.

Chynoweth, D.P., Owens, J., O'Keefe, D., Earle, J.F.K., Bosch, G., Legrand, R., 1992. Sequential batch anaerobic composting of the organic fraction of municipal solid waste. Water Science and Technology 25, 327–339.

Farquhar, G.J., Rovers, F.A., 1973. Gas production during refuse decomposition. Water Air Soil Pollution 2, 483–495.

Kaplovsky, A.J., 1951. Volatile acid production during the digestion of seeded, unseeded and limed fresh solids. Sewage and Industrial Wastes 23, 713–721.

Leckie, J.O., Pacey, J.G., Halvadakis, C., 1979. Landfill management with moisture control. Journal of Environmental Engineering Division ASCE 2, 337–355.

McCarty, P L., 1964a. Anaerobic waste treatment fundamentals. Part 1. Public Works 9, 107–112.

McCarty, P L., 1964b. Anaerobic waste treatment fundamentals. Part 2. Public Works 10, 123–126.

McCarty, P L., 1964c. Anaerobic waste treatment fundamentals. Part 3. Public Works 11, 91–94.

McCarty, P L., 1964d. Anaerobic waste treatment fundamentals. Part 4. Public Works 12, 95–99.

Owens, J., Chynoweth, D.P., 1993. Biochemical methane potential of municipal solid waste (MSW) components. Water Science and Technology 27, 1–14.

Owen, W.F., Stuckey, D.C., Healy, J.B. Jr., Young, L.Y., McCarty, P.L., 1979. Bioassay for monitoring biochemical methane potential and anaerobic toxicity. Water Research 13, 485–492.

Pohland, F.G., Kang, S.J., 1974. Sanitary landfill stabilisation with leachate recycle and residual treatment. In: AIChE Symposium Series. Water, II: Municipal Waste Treatment, pp. 308–327.

Rees, J.F., 1980. The fate of carbon compounds in the landfill disposal of organic matter. Journal of Chemical Technology 30, 161–175.

Robinson, H.D., Maris, P.J., 1979. Leachate from domestic waste: generation, composition and treatment. A review. Water Research Centre, UK, Technical Report TR 108.

Rodhe, H., 1990. A comparison of the contribution of various gases to the greenhouse effect. Science 248, 1217–1219.

Rovers, F.A., Farquhar, G.J., 1973. Infiltration and landfill behaviour. Journal of the Environmental Engineering Division ASCE 99, 671–690.

Stegmann, R., 1983. New aspects on enhancing biological processes in sanitary landfills. Waste Management Research 5, 201–211.

WPCF, 1987. Anaerobic sludge digestion. In:Manual of Practice No. 19, 2nd ed. Water Pollution Control Federation, Alexandra, VA, USA.

Wujcik, W., Jewell, W., 1980. Dry anaerobic fermentation. Biotechnology and Bioengineering Symposium 10, 43–65.