An assessment of the effects of the dual co-disposal of phenol and waste activated sewage sludge with refuse on the refuse anaerobic fermentation and leachate quality

LJ Percival and E Senior*

International Centre of Waste Technology (Africa), University of Natal, Private Bag X01, Scottsville 3209, Pietermaritzburg, South Africa

Abstract

Co-disposal with refuse into controlled landfill sites is the cheapest option for the disposal of hazardous wastes and, if carefully controlled, can be an effective treatment option. In this study a high-strength phenolic waste water together with thickened/dewatered waste activated sewage sludge were co-disposed with refuse. The efficacy of phenol catabolism was assessed in the presence of various co-disposal strategies and leachate recycle was found to facilitate the highest rate. In contrast, batch mode- and single elution-operated microcosms adversely affected the refuse fermentation and, subsequently, leachate quality.

Introduction

The disposal of phenol, one of the most widely produced industrial wastewaters, and of waste activated sludge is of major global concern. Many options are available for the treatment of high-strength phenolic wastewaters although co-disposal in H:h landfills is still the most cost-effective even though liner costs are incurred. This, therefore, commends co-disposal as a viable treatment option for phenolic wastewaters in South Africa.

Waste activated sludge was considered for co-disposal purposes due to the wide use of this sewage treatment process in South Africa. Disposal of waste activated sludge to landfill would reduce the amount requiring treatment by anaerobic digestion, thereby reducing costs. Its overall organic and chemical compositions are not significantly different from other sludge types which are considered more suitable for co-disposal (Sinclair, 1994).

The effects of co-disposing sewage sludge (Blakey, 1991; Beker and Van den Berg, 1991; Watson-Craik et al., 1992; Sinclair, 1994) and phenolic waste waters (Fedorak and Hrudey, 1984; Watson-Craik and Senior, 1990) with refuse have been discussed. According to the UK DOE (1994), if co-disposal is effectively controlled, the leachate produced should not differ greatly from landfill leachate from H:h sites.

The principal aim of this study was to examine for the first time the dual co-disposal of waste activated sludge and phenol with refuse, in the presence of different operational strategies, with specific reference to phenol catabolism, refuse fermentation and leachate quality.

Materials and methods

Refuse

Approximately one-month-old refuse was collected from the Pietermaritzburg landfill site, and hand-sorted to remove all visible glass, metal and plastic. After homogenising in a garden blender, the refuse was stored at 4°C until use.

Sewage sludge

Waste activated sewage sludge was collected from the Darvill Sewage Works from the internal recycle channel of the treatment plant. The sludge was stored at 4°C to allow settlement of the solids. After 5 h the supernatant was discarded. The sludge was thickened further by centrifugation at 1 000 $r \cdot min^{-1}$ x g for 5 min. The supernatant was again discarded. To determine the dry solids content, two samples (100 g) were dried at 70°C for 72 h and then weighed.

Refuse microcosms

Nine glass columns (length 50 cm, i.d. 5.5 cm) were packed and operated as shown in Table 1.

The columns were packed to a fixed refuse density of 750 $kg \cdot m^{-3}$ and a maximum refuse:dewatered sludge ratio of 4:1 (Craft and Blakey, 1988). The columns were connected to zinc acetate (1% w/v) gas traps, to trap generated hydrogen sulphide as zinc sulphide, and were incubated in the dark at 30°C in a temperature-controlled cabinet fitted with heating elements. An empty bed dilution rate (Ismatec peristaltic pump, Type IPN 24B) of 0.026 h^{-1} ± 0.005 h^{-1} was maintained in the single elution columns in accordance with Watson-Craik (1987) and Sinclair (1994). For the leachate recycle and batch microcosms, these were perfused, in the same mode as the single elution columns, with the appropriate phenol concentration for 48 h prior to closure and disconnection of the pump. With the leachate recycle microcosms leachate was drawn off, two or three times per week, from the base of each column, with 10 ml syringes fitted with hypodermic needles, and reintroduced to the top while maintaining anaerobiosis.

After 26 weeks, basic mineral salts medium (Coutts et al., 1984) was added to single elution columns 1A and 1C while Microcosm 1D was resupplemented with phenol (1 000 $mg \cdot l^{-1}$). Columns 1A to 1E and 2A to 2D were monitored for 56 and 23 weeks, respectively.

* To whom all correspondence should be addressed
☎(0331) 260 5527; fax (0331) 260 5527; e-mail seniorm@micr.unp.ac.za
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TABLE 1
PERFUSION REGIMES, MICROCOSM PACKINGS AND ELUENT PHENOL ORGANIC LOADINGS

Microcosm	Perfusion regime	Column packing	Eluent
1A (control)	Single elution	Refuse	Phenol (1 000 mg·l^{-1})
1B (control)	Single elution	Refuse/sludge	Distilled water
1C	Single elution	Refuse/sludge	Phenol (1 000 mg·l^{-1})
1D	Leachate recycle	Refuse/sludge	Phenol (1 000 mg·l^{-1})
1E	Batch	Refuse/sludge	Phenol (1 000 mg·l^{-1})
2A (control)	Single elution	Refuse	Phenol (500 mg·l^{-1})
2B	Single elution	Refuse/sludge	Phenol (500 mg·l^{-1})
2C	Leachate recycle	Refuse/sludge	Phenol (500 mg·l^{-1})
2D	Batch	Refuse/sludge	Phenol (500 mg·l^{-1})

Analytical methods

Volatile fatty acids (VFAs) and methane

VFAs and methane samples were quantified with a Varian 3600 gas chromatograph, equipped with a flame ionisation detector, in which the flow rate of the oxygen-free nitrogen (OFN) carrier gas was maintained at 30 ml·min^{-1}. For VFA analysis a stainless steel column (length 2m, i.d. 4mm) packed with 5% neopentyl glycol sebacate + 1% H_3PO_4 on Anakrom polyester (mesh 80 to 100) was used. The injector and detector temperatures were 200°C and 220°C, respectively. The oven temperature was initially held at 100°C for 2 min, then programmed to increase to 160°C at a ramp rate of 7°C min^{-1}. Formic acid-acidified (1% $^{v}/_{v}$) standards (500, 1 000 and 2 000 mg·l^{-1}) were injected (1 µl) and the concentrations of VFAs calculated by peak area comparison. For methane analysis a glass column (length 1.45 m, i.d. 3 mm) packed with Poropak T (80/100 mesh) was used. The injector, detector and column temperatures were maintained at 110°C, 200°C and 35°C, respectively. The concentrations were calculated by comparison of peak area response to those of standards prepared with pure methane (Fedgas).

Anions

Anions were measured by ion liquid chromatography (ILC) with a Model 430 conductivity detector connected to a Waters 590 programmable pump.

The sodium borate/gluconate concentrate contained the following (g·l^{-1} glass-distilled water): sodium gluconate, 16; boric acid, 18; sodium tetraborate decahydrate, 25. The sodium borate/gluconate concentrate eluent, with a conductivity of approximately 270 µS, contained the following (mg·ml^{-1} glass-distilled water): borate/gluconate concentrate, 20; acetonitrile, 120.

Samples (100 µl) were injected into an IC-Pak A column (4.6 x 50mm) which contained trimethylammonium functionalised polymethacrylate, water, lithium meta-borate and sodium gluconate (10 µm particle size). Standards of nitrite, nitrate, phosphate and sulphate (5, 5, 10, 5 mg·l^{-1}, respectively) were used.

Ammonia

Ammonia was measured with an Orion Model 95-12 ammonia electrode connected to an Orion Research Model 701/A digital ionalyser.

pH

pH was measured with a Crison MicropH2000 pH meter.

Results and discussion

In this study, due to the number of variables examined, laboratory-scale microcosms were employed rather than lysimeters with working volumes of 0.5 to 1 m^3.

The moisture content of the refuse used in the microcosms was 60.8% ± 1.6 (w/w) and was, thus, close to field capacity. The resulting solids content of the waste activated sewage sludge, after centrifugation, was 25% (w/v) which was equivalent to a dry solids content of 4.95% ± 0.5 (w/w). The refuse was not saturated prior to perfusion with phenol to approximate on-site conditions.

In the three dual co-disposal microcosms operated with a single elution regime (1B, 1C and 2B), "ponding" occurred. No "ponding" was, however, observed in the equivalent controls (1A and 2A) which suggested that the waste activated sludge was restricting liquid infiltration.

Phenol catabolism

Figures 1A-H show the changes in the leachate residual phenol concentrations of the microcosms challenged with this electron donor.

In the single elution columns relatively ineffective phenol removals were recorded. During 56 weeks of operation the total amounts of influent phenol which were removed by passage through Microcosms 1A and 1C were 10.8% and 14.0%, respectively (Figs. 1A and 1B). For these columns, the addition of nutrients at Week 26 did not promote phenol catabolism which suggested that the refuse was not nutrient-limited. For the same microcosms, the phenol removals during the initial 23 weeks of the study were 8.8% and 14%, respectively. In comparison, single elution columns 2A and 2B facilitated total phenol concentration reductions of 26.2% and 18.2%, respectively (Figs. 1C and 1D).

A number of factors could account for the limited effectiveness of the single elution columns to catabolise phenol. Channelling, in part facilitated by the walls of the small microcosms, may have occurred allowing the phenol solution to move rapidly through the refuse (Watson-Craik and Senior, 1989) thus reducing the contact time between the microorganisms present and the phenol. VFAs may also have been removed by the same liquid flux thus protracting the time taken to reach a balanced fermen-

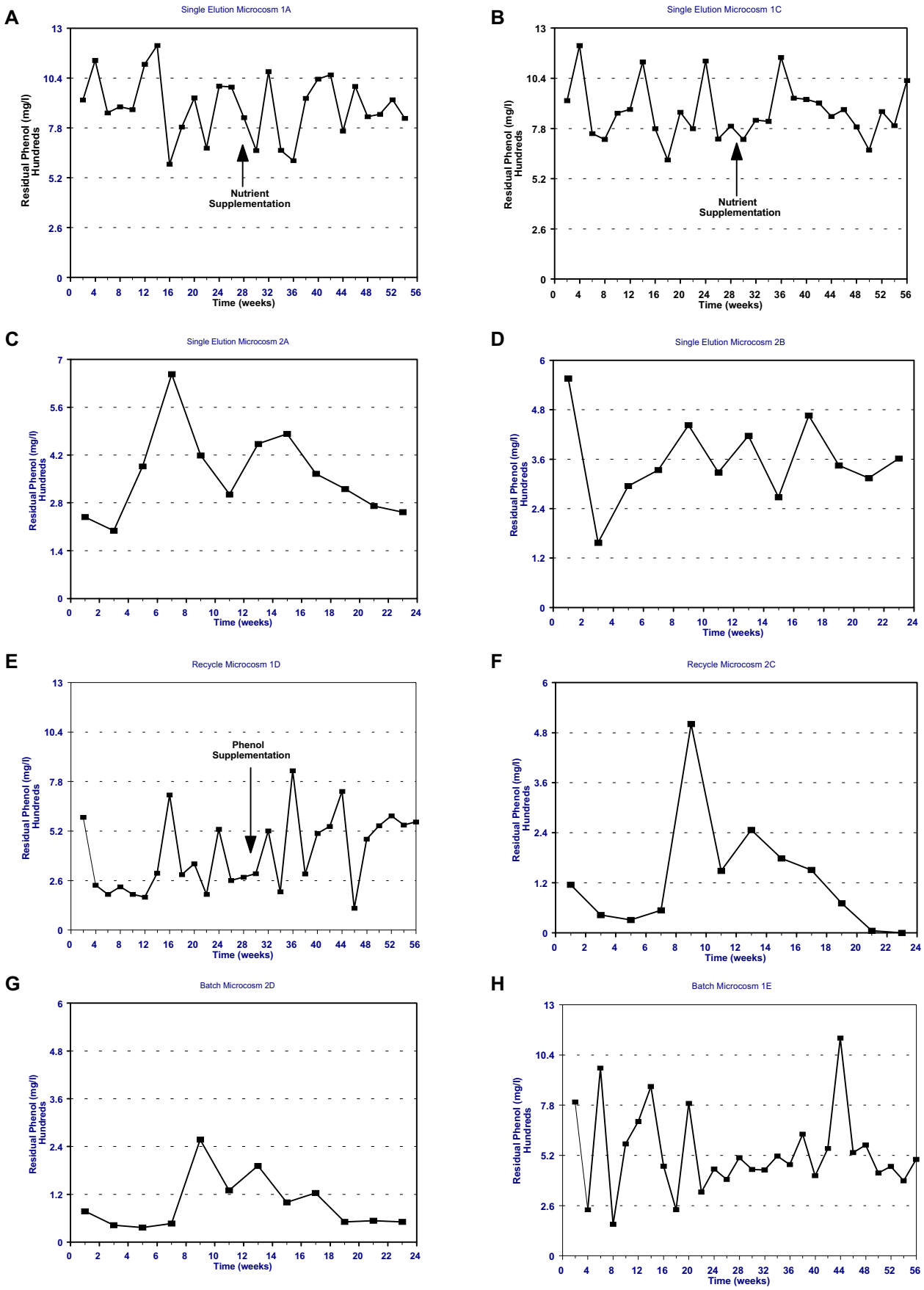


Figure 1

*Leachate residual phenol concentrations of Microcosms 1A (**A**), 1C (**B**), 2A (**C**), 2B (**D**), 1D (**E**), 2C (**F**), 2D (**G**),* and 1E *(**H**)*

tation. It is also possible that phenol toxicity may have been operative. Previous studies with a multi-stage chemostat have demonstrated that 753 mg·l^{-1} phenol inhibited phenol catabolism (Watson-Craik and Senior, 1990). This could explain the slightly higher phenol removal efficiencies of Microcosms 2A and 2B which were perfused with the lower (500 mg·l^{-1}) phenol concentration.

The temporal changes in the leachate residual phenol concentrations of the single elution microcosms were possibly due to reversible adsorption on the refuse components. The bonds between phenol molecules and refuse components are relatively weak and are easily broken (desorption) by continuous liquid infiltration (Costley, 1995).

With the exception of Microcosm 1E, initially, low concentrations of residual phenol were recorded with the microcosms operated with batch or leachate recycle (Figs. 1E-1G). These low concentrations were attributed to adsorption on the refuse components or, less likely, to the presence of aerobic or facultative anaerobic phenol degraders. In Microcosm 2C, recycling 500 mg·l^{-1} phenol resulted in the establishment of a metabolically active phenol-degrading microbial population and no residual phenol was detected by Week 23. A similar removal was not, however, apparent with the higher phenol organic loading rate (Microcosm 1D, Fig. 1E).

For batch-operated Microcosm 1E there was no significant reduction in the initial residual phenol concentration compared with the other batch and recirculation columns. This was attributed to low pH-mediated desorption. Also, it is possible that the low pH increased the bactericidal/bacteriostatic affect of the phenol. As expected, the 500 mg·l^{-1} phenol-supplemented batch fermentation (Microcosm 2D) effected a greater concentration reduction than the equivalent 1 000 mg·l^{-1}-supplemented microcosm (1E) for the same 23 weeks (Figs. 1G and 1H).

Refuse fermentation parameters

Volatile fatty acids

In this paper, the term VFAs relates to the volatile fatty acids (sometimes referred to as short-chain fatty acids or SCFA) and their corresponding salts.

In all single elution microcosms there was an initial flush of VFAs (Figs. 2A-2E). In the columns perfused with phenol (Microcosms 1A, 1C, 2A and 2B), as expected, acetate predominated. The phenol had no apparent inhibitory effect on the production/release of VFAs from the refuse as comparable concentrations were eluted from the control (Microcosm 1B, Fig. 2E) and the single elution microcosms. The one observed difference was the initially higher propionate concentration compared to the acetate concentration of Microcosm 1B. Previously, Sinclair (1994) observed an apparent stimulatory effect on propionate generation in the presence of a refuse:dewatered sludge ratio < 4.1:1 compared to a refuse control following perfusion with water.

The initially high VFA concentrations in the two columns operated with leachate recycle (Figs. 2F and 2G) were higher than in corresponding batch fermentations (Figs. 2H and 2I) and suggested that recycling promoted the establishment of an acidogenic population (and effected VFA accumulation) despite the presence of phenol. The high acetate concentrations recorded in these columns possibly explained the ineffectiveness of phenol degradation since it has been reported that low cultural concentrations were required for phenol catabolism (Senior and Balba, 1990). In Microcosm 2C, all the VFAs had been attenuated by Week 23 (Fig. 2G) which coincided with a zero residual phenol concentration.

Microcosm 1E (Fig. 2H) was characterised by high concentrations of VFAs which suggested that acidogenesis exceeded acidotrophy. It is possible that the addition of waste activated sludge promoted the development of this active acidogenic population. Unfortunately, a batch fermentation control was not included in the study so this possible promotion could not be studied further. Previous work by Fedorak and Hrudey (1984) demonstrated inhibition of acidogenesis in batch fermentation studies in the presence of 2 000 mg·l^{-1} phenol. Although this concentration would be high for a landfill, the result suggested that in our study phenol was not an inhibitory factor in the batch microcosms. Whilst not recorded, it is possible that the pH values of the batch fermentations of Fedorak and Hrudey (1984) were higher than those recorded in this study. The pH in Microcosm 1E (Fig. 3H) was exceptionally low and may have increased the bactericidal effect of the phenol (Karabit et al., 1985) and, thus, reduced the concentration required to inhibit acidogenesis.

The branched-chain fatty acid *iso*valeric acid was detected, for the duration of the study, only in the microcosms operated with recycle or batch regimes which indicated that protein deamination had occurred.

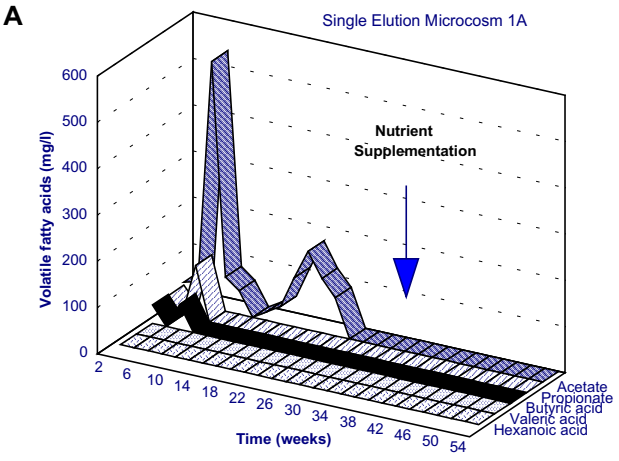
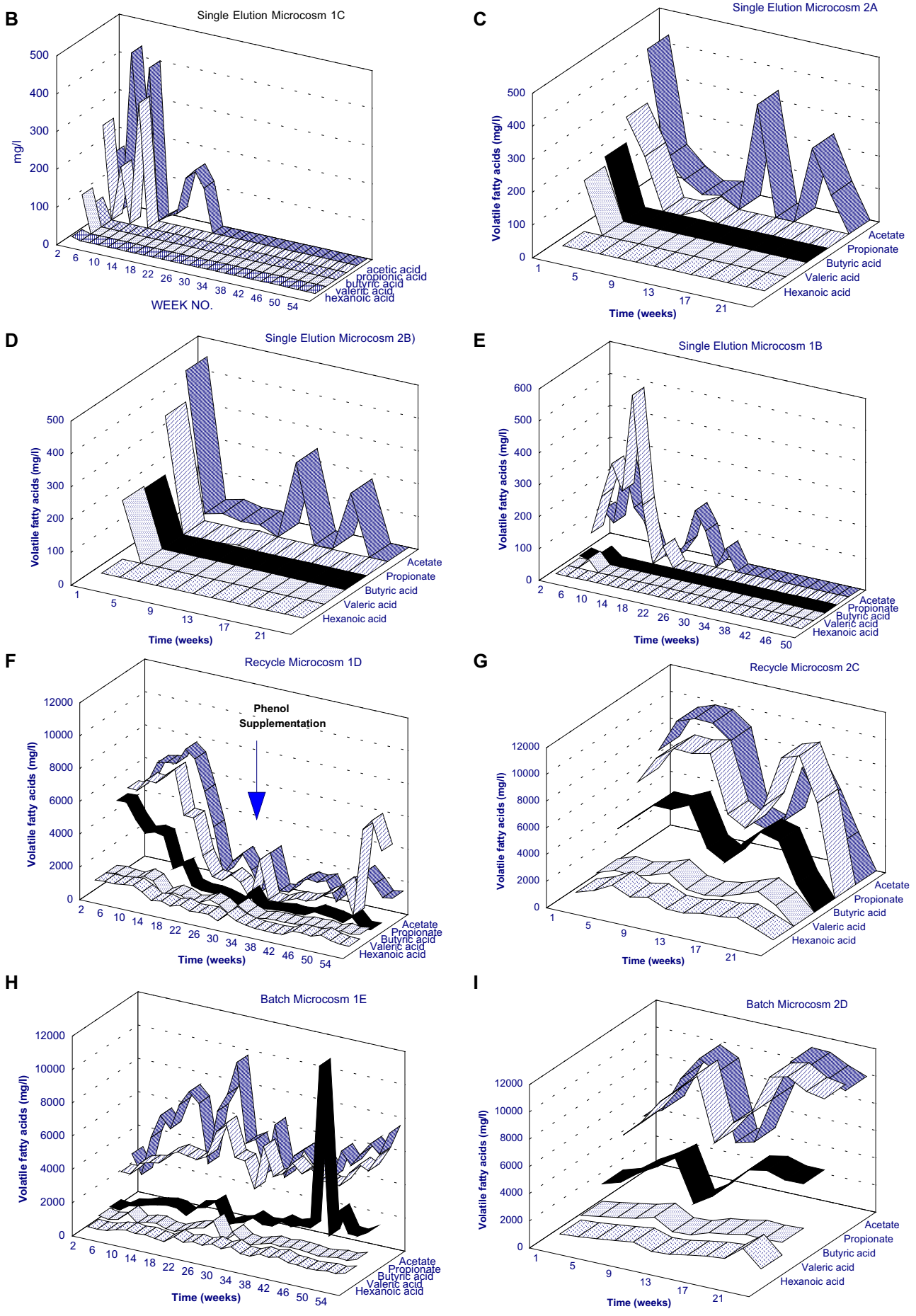


Figure 2
*Leachate VFA concentrations of Microcosms 1A (**A**), 1C (**B**), 2A (**C**), 2B (**D**), 1B (**E**), 1D (**F**), 2C (**G**), 1E (**H**) and 2D (**I**)*


B
Single Elution Microcosm 1C
mg/l
WEEK NO.
acetic acid
propionic acid
butyric acid
valeric acid
hexanoic acid
C
Single Elution Microcosm 2A
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
D
Single Elution Microcosm 2B)
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
E
Single Elution Microcosm 1B
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
F
Recycle Microcosm 1D
Phenol
Supplementation
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
G
Recycle Microcosm 2C
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
H
Batch Microcosm 1E
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid
I
Batch Microcosm 2D
Volatile fatty acids (mg/l)
Time (weeks)
Acetate
Propionate
Butyric acid
Valeric acid
Hexanoic acid

pH

pH is an important parameter to monitor during co-disposal practice, particularly with phenolic waste waters, as it controls the toxic effect of phenol and the possible inhibition of methanogenesis.

Concomitant with the lower concentrations of VFAs, the pH values of the single-elution microcosm leachates (1A, 1C, 2A and 2B) were higher throughout the whole study than the corresponding leachate recycle and batch fermentation microcosms (Figs. 3A to 3I). Sludge addition had no obvious effect on the initial pH.

With Microcosm 1B the leachate pH progressively increased during the 56 weeks of study to >6 (Fig. 3E). This rise, in conjunction with an absence of VFAs (Fig. 2E), suggested that methanogenesis had established and acidotrophy was proceeding faster than acidogenesis.

In the presence of 1 000 mg·l^{-1} phenol recycle (Microcosm 1D), the leachate pH fluctuated about a mean of 4.6. This decreased pH coincided with high VFA concentrations. The substantial increase in the leachate pH recorded after 22 weeks in the corresponding lower phenol-loaded microcosm (2C) (Fig. 3G) coincided with a reduction in the phenol (Fig. 1F) and VFA (Fig. 2G) concentrations.

As a consequence of the high concentration of VFAs produced during the batch fermentations the pH values (Figs. 3H and 3I) were much lower than in the corresponding single elution and recycle columns. For the duration of the experiment the pH values never increased, which confirmed that "souring" (acid accumulation) had occurred in both columns.

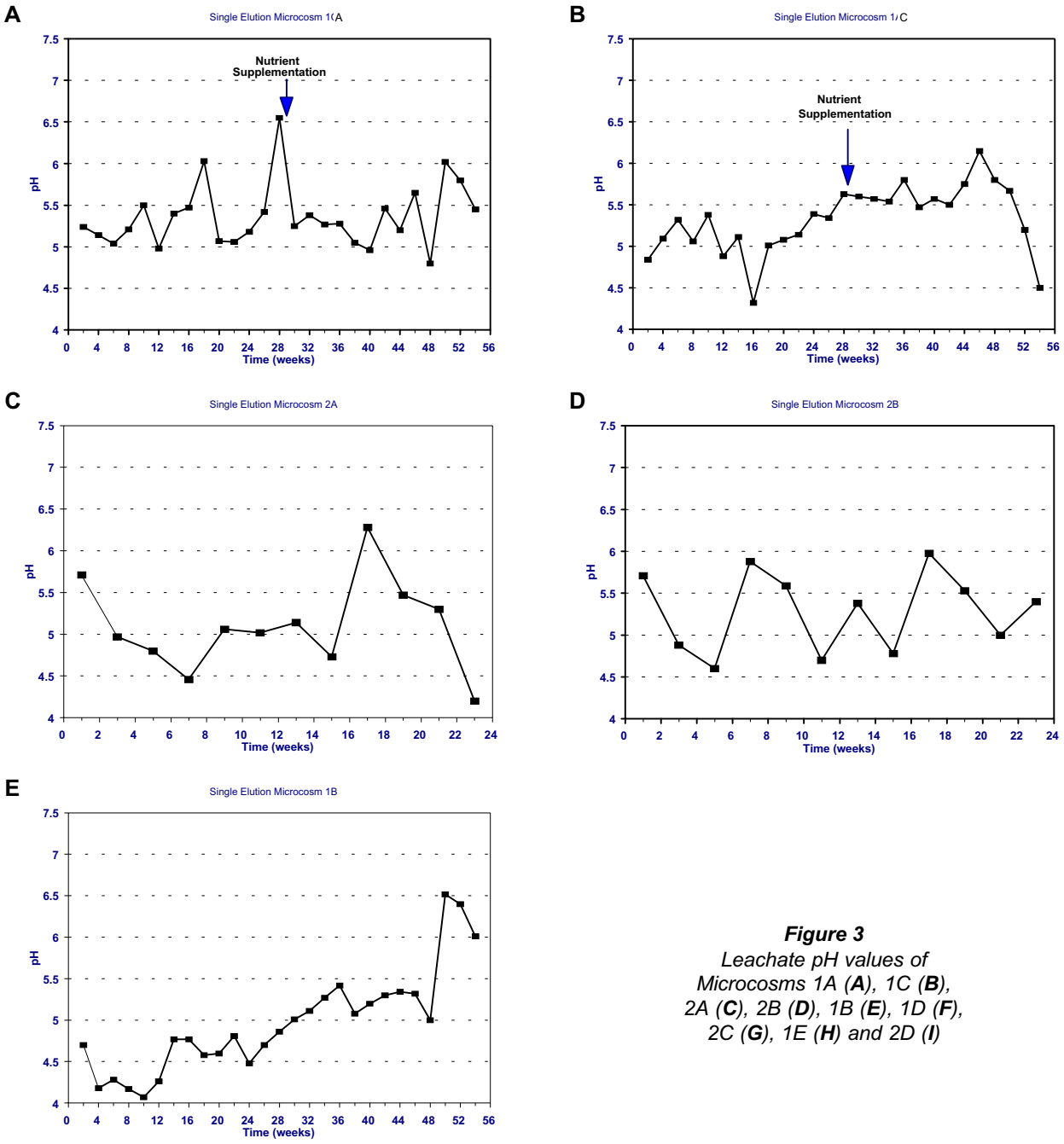
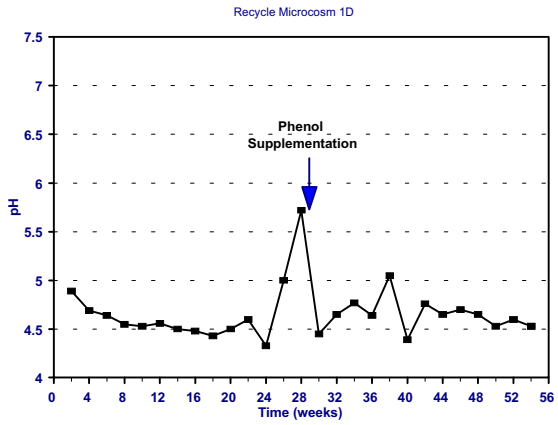
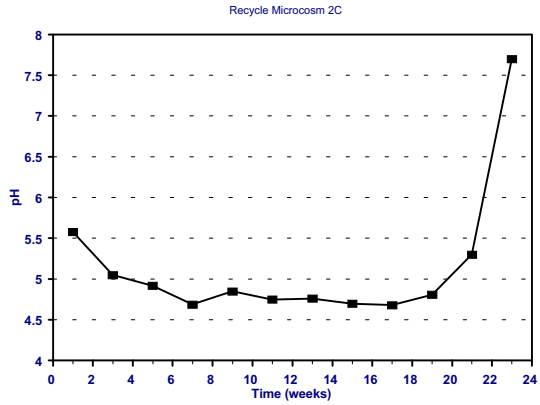


Figure 3
*Leachate pH values of Microcosms 1A (**A**), 1C (**B**), 2A (**C**), 2B (**D**), 1B (**E**), 1D (**F**), 2C (**G**), 1E (**H**) and 2D (**I**)*

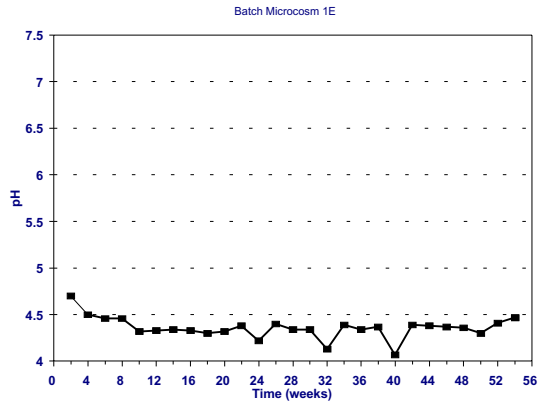
F



G



H



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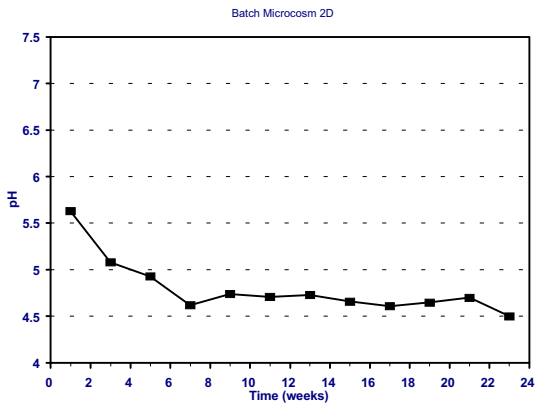


Sulphate

Monitoring the sulphate concentration in landfill leachate can determine the development of a sulphate- reducing bacterial population and the possible inhibition of the methanogens. The leachate sulphate concentrations were very low in control Microcosm 1B for the duration of the study (Fig. 4A) and suggested effective removal by sulphate-reducing bacteria.

The sulphate concentrations of Microcosm 1C leachate (Fig. 4B) were generally comparable with Microcosm 1A for the duration of the study.

The highest initial leachate sulphate concentrations (>230 $mg \cdot l^{-1}$) were recorded with Microcosms 2A and 2B (Figs. 4D and 4E). These concentrations rapidly decreased, however, possibly due to washout.

With Microcosm 2C, leachate recycle effected a black precipitate at the top of the microcosm. This suggested the evolution of hydrogen sulphide which forms during dissimilatory sulphate reduction (Widdel, 1988) when sulphate concentrations are high (Fig. 4F). Sulphate reduction to hydrogen sulphide is energetically more favourable than methane production from both hydrogen/carbon dioxide and acetate (Zeikus, 1983) so it would be expected that methanogenesis would be inhibited, although analysis of the headspace gas indicated that this was not the case (Table 2). Blackening was never observed in the recycle Microcosm 1D, possibly due to the low sulphate concentrations detected in the leachate (Fig. 4G).

Sulphate-reducing bacteria have been shown to be more susceptible to elevated organic loads than methanogens with 6 000 $mg \cdot l^{-1}$ acetate and 7 300 $mg \cdot l^{-1}$ propionate inhibiting the former (Watson-Craik et al., 1993). Considering this, and the fact that sulphate-reducing bacteria are usually inhibited at pH <6 (Widdel, 1988), these bacteria should have been inhibited in the batch-operated microcosms (1E and 2D). However, the low sulphate concentrations detected (Figs. 4H and 4I) suggested that this was not the case. It is possible that the sulphate-reducing bacteria may have been active in the microcosms which developed low pH values by occupying microniches in the refuse where buffering effected higher pH values (Widdel, 1988). The sulphate concentration did, however, increase dramatically after Week 14 in Microcosm 2D which suggested less sulphate-reducing activity and, therefore, less competition for hydrogen/acetate with the methanogens. However, this possibility was not supported by the methane concentration data.

Methane

Previous studies have demonstrated increased methane production following anaerobic sewage sludge co-disposal with refuse (Blakey, 1991; Beker and Van den Berg, 1991).

In this present study, with the exception of Microcosm 1A, methane was evolved from all the columns. Although unlikely, the initial flush of methane from Microcosm 1C (Fig. 5A) may have been from the sludge since no methane was evolved from Microcosm 1A. Confirmatory evidence was not, however, gained with control Microcosm 2A since methane concentrations as high as 48.4% ($^{v}/_{v}$) were recorded (Table 2). The other results obtained with the microcosms perfused with 500 $mg \cdot l^{-1}$ phenol (Table 2) were inconsistent and could not be used to compare the two phenol organic loadings. Methane was not detected in Microcosm 1C between Weeks 8 and 24. This was possibly due to phenol toxicity since phenol has been reported to be inhibitory to methanogens in concentrations $\geq$753 $mg \cdot l^{-1}$ (Watson-Craik,

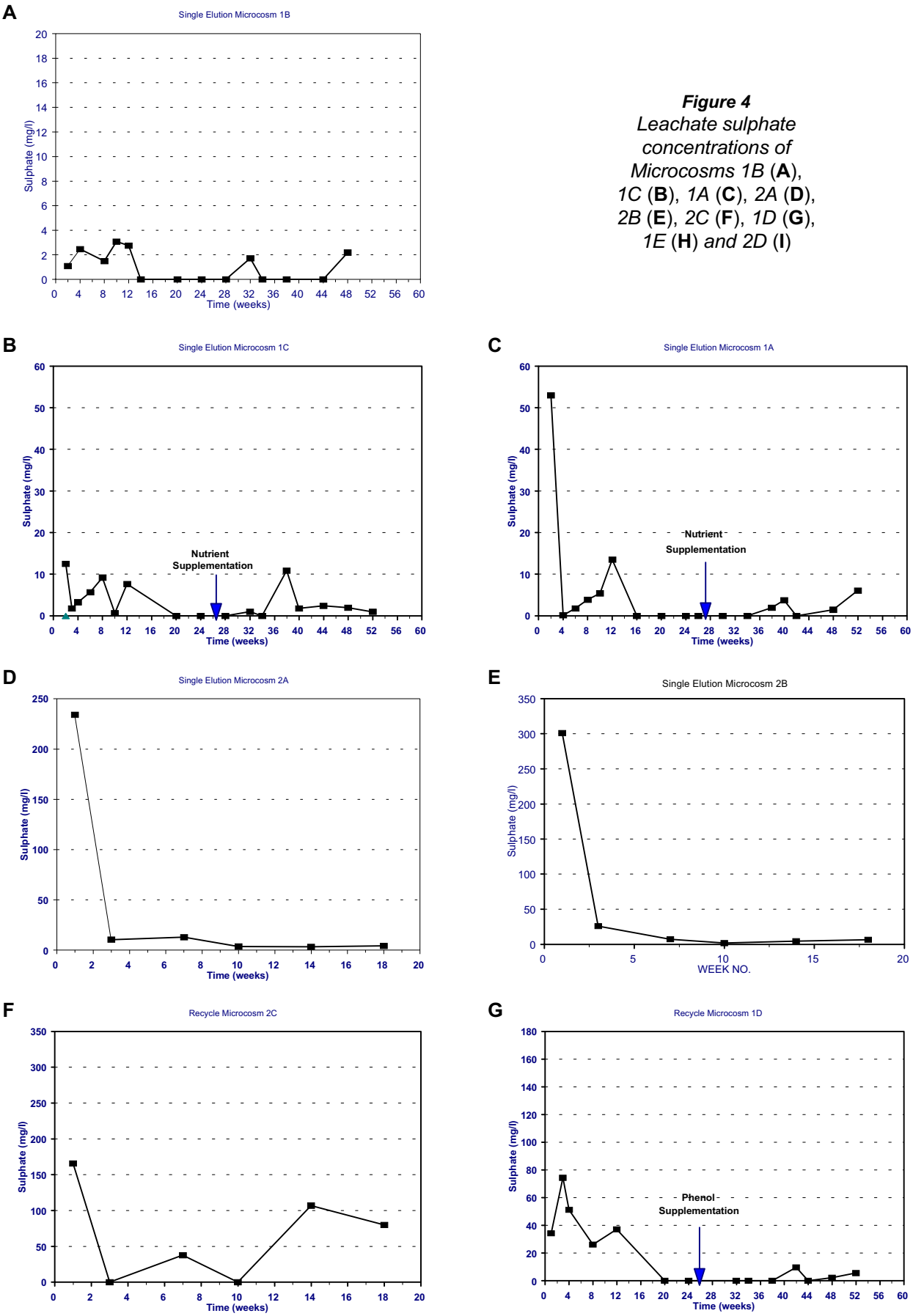
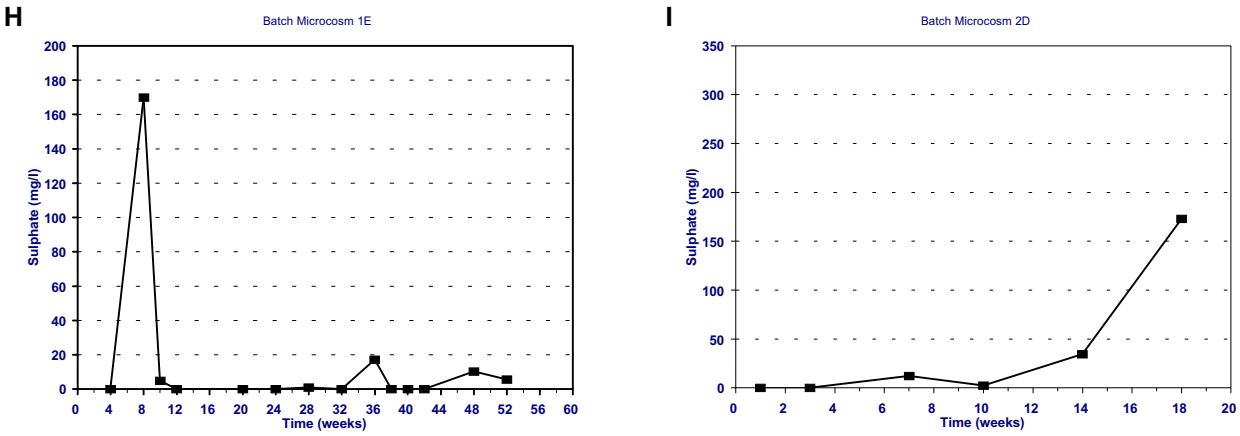


Figure 4
Leachate sulphate concentrations of Microcosms 1B (**A**), *1C* (**B**), *1A* (**C**), *2A* (**D**), *2B* (**E**), *2C* (**F**), *1D* (**G**), *1E* (**H**) *and 2D* (**I**)



1987). The methane concentration increase after Week 24 confirmed the establishment of a methanogenic population. By comparison, in Control Microcosm 1B, headspace methane was detected after 8 weeks although it took >18 weeks before increased methanogenesis was recorded (Fig. 5B).

The highest methane concentration was detected in Microcosm 1D after 60 weeks of operation (Fig. 5C). This was as expected since recycling leachate has been reported to reduce the lag phase of methanogenesis due to recirculation of acetate, a major precursor of methanogenesis.

Methane was evolved in batch fermentation 1E soon after packing (Fig. 5D) despite the "souring" of the column and reported inhibition by phenol concentrations $\geq$753 mg·l^{-1}. Previous studies have reported methanogenesis at pH values $\geq$5 with establishment of an active methanogenic population attributed to the presence of methanogens in refuse microniches (Watson-Craik, 1987). With this microcosm, the concentration of methane decreased gradually during the study due, possibly, to the loss of buffered microniches within the refuse and subsequent inhibition by high concentrations of VFAs (Fig. 2H) and, subsequent, low pH values (Fig. 3H). A similar reduction in methane concentration was also recorded with Microcosm 2D at Week 20 (Table 2).

TABLE 2
PERCENTAGE (V/V) METHANE DETECTED IN THE HEADSPACES OF MICROCOSMS 2A - 2D

Week number	Microcosm 2A	Microcosm 2B	Microcosm 2C	Microcosm 2D
10	48.4	9.7	13.1	11.6
20	1.2	N/A	18.3	4.3

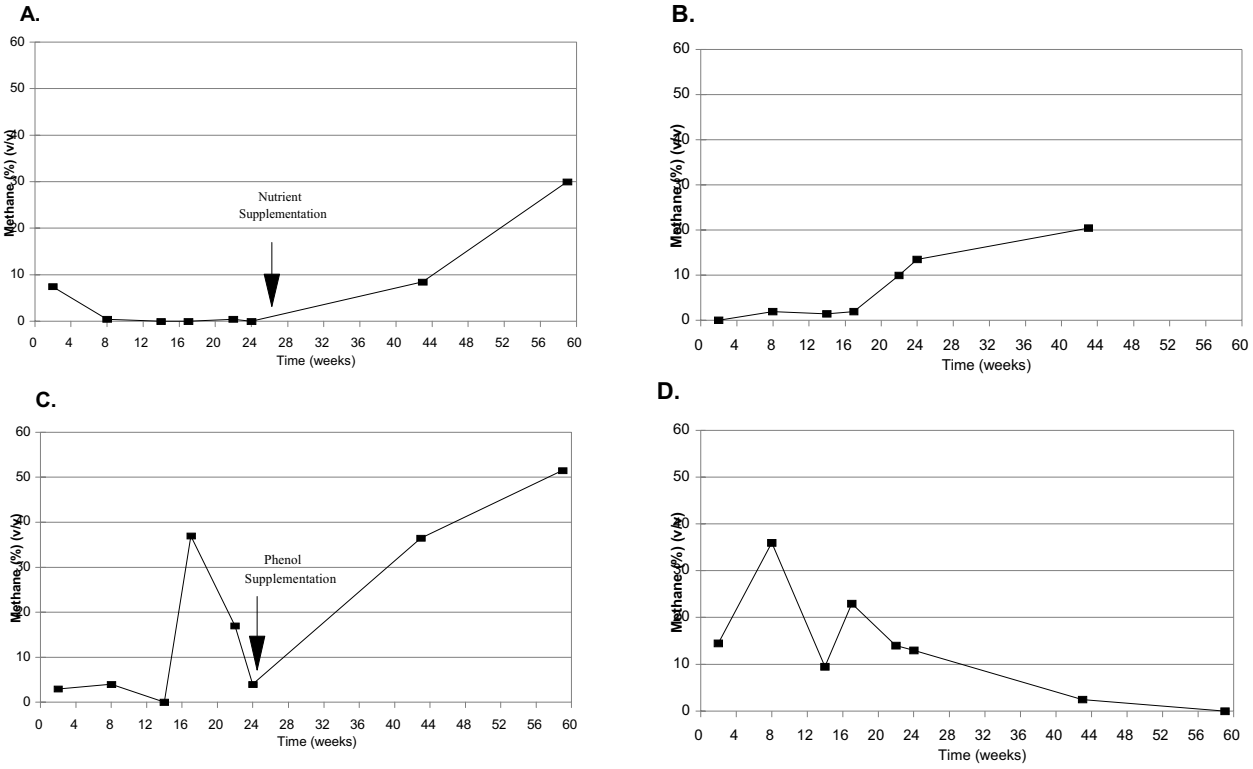


Figure 5
*Headspace methane concentrations of Microcosms 1C (**A**), 1B (**B**), 1D (**C**) and 1E (**D**)*

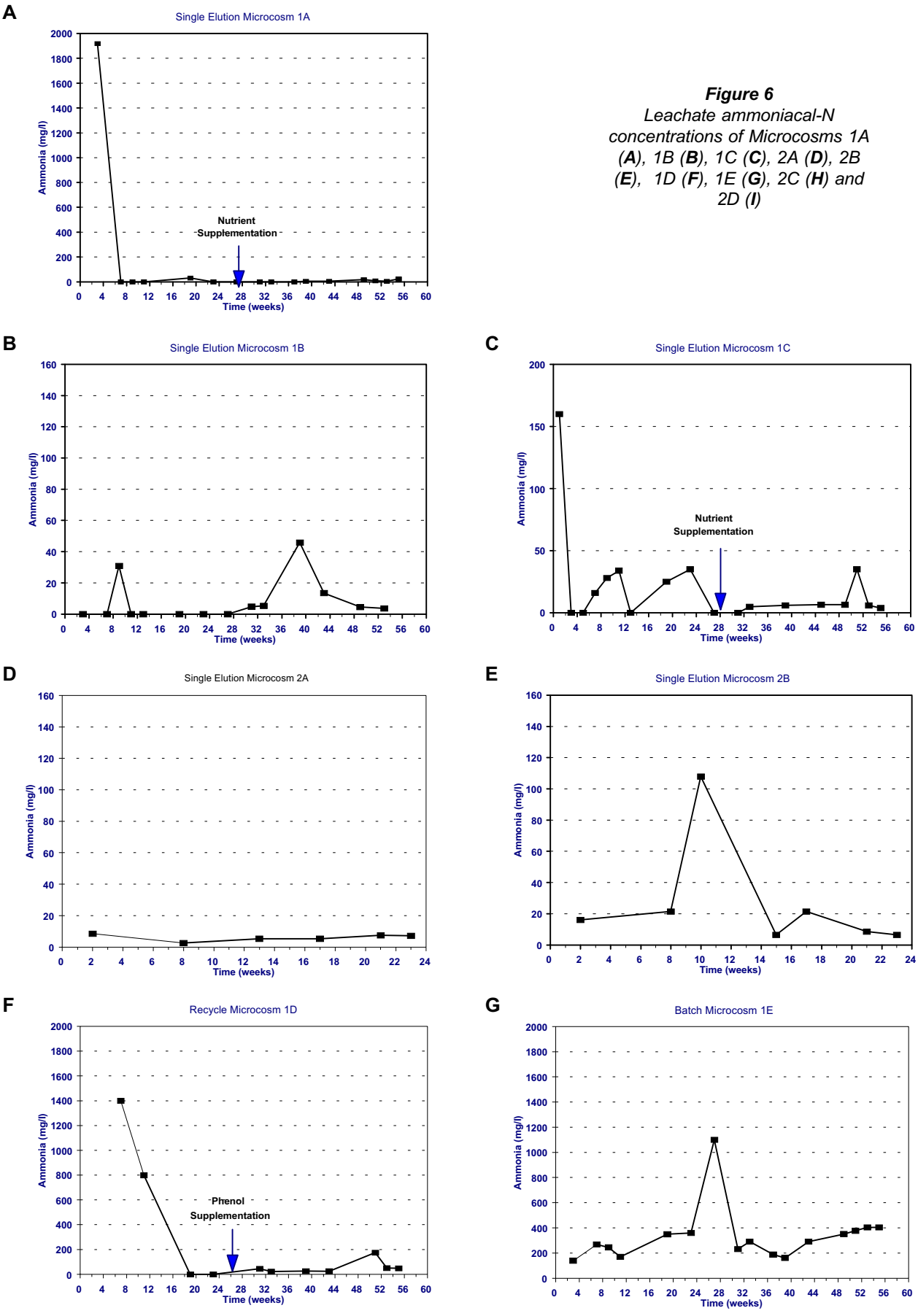
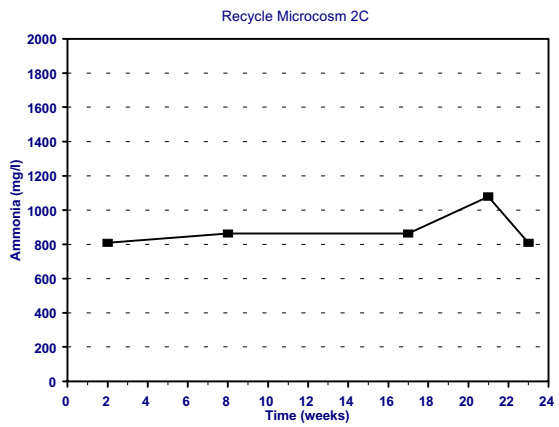


Figure 6
*Leachate ammoniacal-N concentrations of Microcosms 1A (**A**), 1B (**B**), 1C (**C**), 2A (**D**), 2B (**E**), 1D (**F**), 1E (**G**), 2C (**H**) and 2D (**I**)*

H



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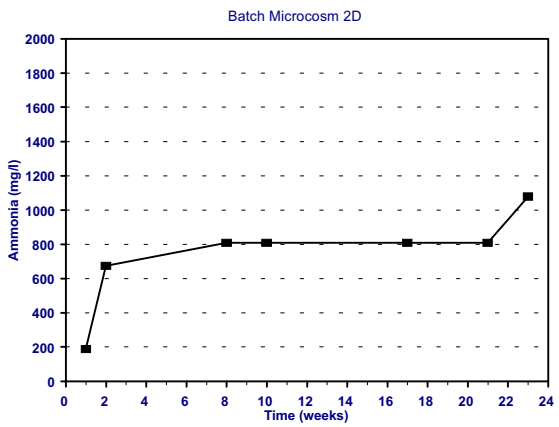


Ammoniacal-N

Co-disposal of sewage sludge in landfill will increase the concentration of nitrogenous compounds in the refuse mass (Watson-Craik et al., 1992). For non-nitrogen-limited growth a C:N ratio <30 is generally considered necessary (UK DOE, 1986) in landfill.

Whilst not monitored in this programme, laboratory- (microcosm) and pilot-scale (lysimeter) studies, unlike landfills, have been characterised by loss of ammonia in the biogas.

The highest initial ammoniacal-N concentration was recorded in control Microcosm 1A which contained refuse only (Fig. 6A), whereas no ammoniacal-N was detected in sludge-amended Microcosm 1B (Fig. 6B). The initial flushes in ammoniacal-N in the leachates of Microcosms 1C, 2B, 1D, 1E, 2C and 2D (Figs. 6C, 6E-6I) could, therefore, not be attributed to the presence of free ammoniacal-N in the sludge. Dissimilatory nitrate reduction was unlikely to have contributed to the initial flushes from the refuse as anaerobic microbial populations would have only been partially developed.

The waste activated sludge may have introduced nitrifying bacteria into the refuse mass. Nitrification in a refuse mass, however, only occurs in areas where there is air ingression. Thus, oxidation of ammonium should only occur for a few weeks following sludge emplacement. It is unlikely that nitrification would have occurred in this study due to the toxic effect of phenol on the nitrifying bacteria. Theoretically, nitrification was only possible in Microcosm 1B. This could possibly account for the absence of ammoniacal-N during the first few weeks of the study (Fig. 6B). In contrast, ammonia was detected during the corresponding first few weeks in all the other microcosms and was attributed to phenol toxicity of the nitrifiers.

In all the single elution columns (1A, 1B, 1C, 2A and 2B) continuous leaching would have prevented accumulation of ammoniacal-N thus explaining the lower leachate concentrations (Figs. 6A-6E). Dissimilatory nitrate reduction, however, possibly explained the presence of ammoniacal-N in the batch and recycle columns (1E, 2D; 1D, 2C) following the initial flushes (Figs. 6F-6I). The concentrations were very high since any ammoniacal-N produced would be retained/recycled instead of being leached out.

Nitrate/Nitrite

In the absence of waste activated sludge co-disposal, the concentrations of nitrate and nitrite in refuse are, characteristically, low. These concentrations are, however, important factors in the attainment of a balanced methanogenic fermentation since denitrification is thermodynamically more favourable than methanogenesis (Oremland, 1988). A refuse mass in the methanogenic phase of activity should, however, be capable of maintaining a denitrifying population.

The detection of nitrate in Microcosm 2C at Week 7 (Fig. 6G) was possibly due to nitrification although this was unlikely owing to the presence of a high phenol concentration. It is also possible that the nitrate was leached from the refuse or sludge emplaced in the column.

The rapid increases in nitrate concentrations at Week 40 in Microcosms 1A and 1C to >300 $mg \cdot l^{-1}$, which coincided with leachate sulphate concentration increases, were difficult to explain. Since production of nitrate only occurs under aerobic or facultatively anaerobic conditions, a possible explanation was the presence of heterotrophic nitrifying bacteria whose growth under conditions of low oxygen and acidic pH has been reported (Beckman et al., 1972).

It was expected that Microcosm 1B would contain higher concentrations of nitrate than those recorded due to the presence of sewage sludge. Similar low concentrations were, however, reported by Sinclair (1994) who suggested that in co-disposal of waste activated sludge with refuse the latter was the main source of leachate nitrate.

The leachate nitrate concentrations of Microcosm 1D were negligible but at Week 40 a high nitrite concentration was detected (Fig. 7F). In the corresponding lower phenol concentration leachate recycle microcosm (2C), a nitrite increase occurred after Week 10, and followed a nitrate peak at Week 7 (Fig. 7G). It is possible that this was due to denitrification although the resulting nitrite concentration was exceptionally high. The presence of phenol could have effected inhibition of nitrite reductase in the denitrification process.

There was no obvious explanation for the high nitrate concentration detected in Microcosm 1E (Fig. 7H) since this column was operated as a closed culture with no air ingression. Subsequently, methanogenesis was not inhibited by the high nitrate concentration which should have diverted electron flow away from the methanogens. The nitrate concentration increase did, however, follow an ammonium concentration increase (Fig. 6G). It is possible, therefore, that during sample storage, prior to analysis, nitrification occurred. This could also have explained the elevated nitrate concentrations recorded with Microcosm 2D (Fig. 7I).

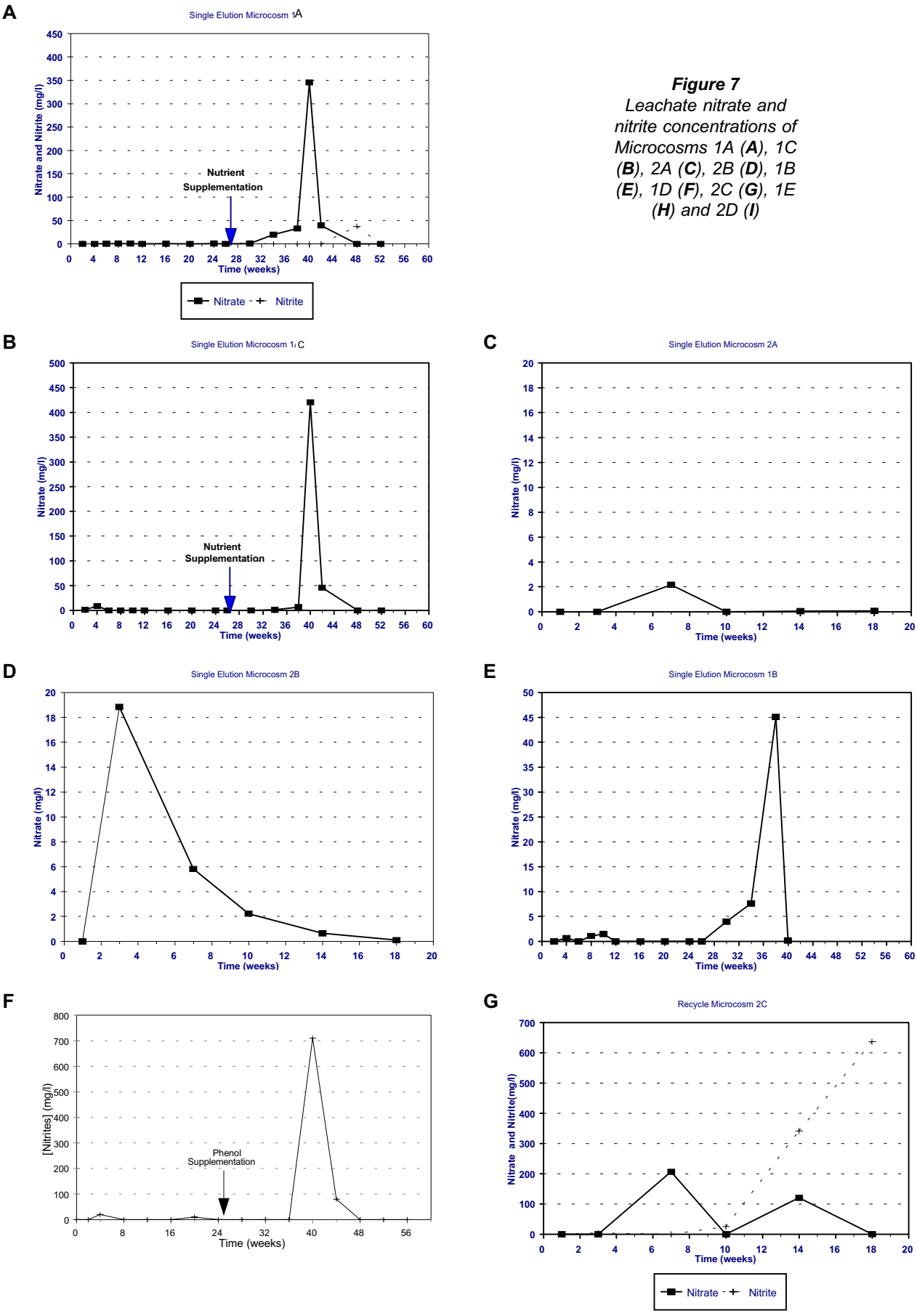
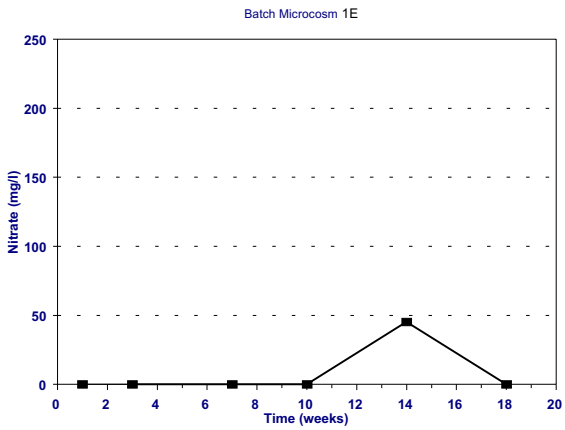
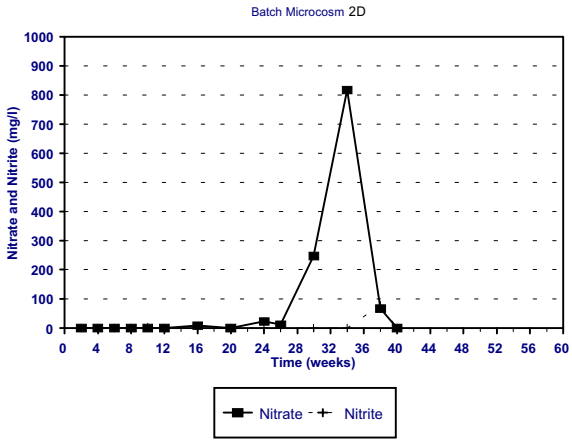


Figure 7
*Leachate nitrate and nitrite concentrations of Microcosms 1A (**A**), 1C (**B**), 2A (**C**), 2B (**D**), 1B (**E**), 1D (**F**), 2C (**G**), 1E (**H**) and 2D (**I**)*

H



I



Conclusion

This present study confirmed that a high concentration of phenol ($\geq$1 000 mg·l^{-1}) has a detrimental effect on phenol catabolism in refuse. Refuse acidogenesis and methanogenesis were not directly inhibited by the co-disposed phenol but depressed pH increased its bactericidal efficacy and limited methanogenesis *per se*. It is, therefore, unwise to co-dispose high-strength phenolic waste water with refuse in the acidogenic phase of degradation, particularly since heavy metal release rather than attenuation is operative in this phase. Co-disposal should be practised either when the refuse is in the methanogenic phase or with refuse situated above methanogenic phase refuse. The results of this study also confirmed that high-strength phenolic waste water should not be co-disposed with "fresh" refuse due to the resulting low pH and slow establishment of an anaerobic microbial population. One possible solution to this problem would be to co-dispose a high-strength phenolic waste water with an alkaline waste such as lime. Tibbles and Baecker (1989) suggested implementation of this practice following a drop in the leachate pH to <6.

Leachate recycle, and, hence, increased retention in the refuse, was necessary to facilitate an effective reduction in the phenol concentration. Time was also critical for the selection/acclimation of a phenol-catabolising microbial population. Unfortunately, as a result of leachate recycle, a high concentration of ammoniacal-N resulted which would otherwise have been progressively released. This could, thus, pose a serious environmental impact if it is not effectively retained within the refuse mass. An increased production of hydrogen sulphide would also be detrimental to landfill gas/leachate quality. Of concern also was the bacteriostatic/bactericidal effect of phenol on the nitrification process. This could cause problems in the aerobic biological treatment of a leachate which contains high concentrations of ammoniacal-N and phenol.

Operation of a co-disposal site as a batch process is impractical. The results in this study, however, gave an insight of the dual co-disposal of phenol and waste activated sludge with refuse in the absence of further liquid infiltration. At the highest phenol loading (1 000 mg·l^{-1}) high VFA concentrations and, subsequently, low pH values resulted in inhibition of the phenol degraders, acidogens and methanogens. Co-disposal of 500 mg·l^{-1} phenol had a lower inimical effect but emphasised the need to determine a maximum organic loading rate.

Co-disposal of waste activated sludge with refuse had no obvious detrimental effects on the refuse fermentation although the presence of sludge did not promote phenol catabolism. For successful co-disposal of activated sludge with refuse, initial dewatering problems and "ponding" in the refuse mass must be addressed. Also, a landfill liner is required in conjunction with a specific leachate management strategy.

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