Treatment of a high-strength leachate from a closed co-disposal landfill site in South Africa

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Abstract

A high-strength leachate from a closed co-disposal landfill site was characterised to determine its chemical composition and susceptibility to biological treatment. The leachate required dilution to 25% (v/v) before it responded to aerobic catabolism. Complete anaerobic treatment was ineffective even with a final dilution of 90% (v/v) of the original leachate. Indirect inhibition of methanogenesis by the high sulphate concentration was the probable cause. Following phosphate addition, aerobic biological treatment effected a significant chemical oxygen demand (COD) reduction but did not lower the ammoniacal-N concentration. Scaling and precipitation occurred which did not adversely affect the biological process but could cause operational problems in full-scale leachate treatment plants. Ion exchange, with soil and lime addition, was, therefore, considered to effect inorganic content reductions prior to biological treatment.

Introduction

Landfill leachate, which originates from water which has percolated through emplaced refuse, is a complex and highly polluting waste water which contains organic and inorganic materials and suspended solids (Ho et al., 1974; Chian and DeWalle, 1976). If leachate is allowed to enter groundwater it can have serious environmental impacts. Protection of groundwater is of particular concern in South Africa. At present, most leachate produced by South African landfill sites is either discharged to sewer or disposed to land. Unfortunately, these practices are often uncontrolled. Treatment of landfill leachate may, therefore, be necessary to minimise the pollution potential.

Complete characterisation of a leachate is a pre-requisite for determining a suitable treatment. Such analysis provides information of the microbiological processes operative within the landfill and identifies the microbiocidal components which may limit biological treatment, or which cannot be discharged to sewer (Chu et al., 1994).

Landfill leachate can be treated *in situ*, by recirculation back through the refuse mass, or can be collected and treated externally by biological and physico-chemical methods. Biological treatment (aerobic and anaerobic), which is generally considered to be reliable, simple and cost-effective, is suitable for leachates which contain high concentrations of volatile fatty acids (VFAs). Reductions of >90% in COD (Chian and DeWalle, 1976; Robinson et al., 1982; Robinson and Maris, 1985) and BOD (Boyle and Ham, 1974) have been observed in laboratory studies. Knox (1985) and Robinson and Luo (1991) also demonstrated ammonia removal through nitrification during aerobic treatment. Physico-chemical treatment is ineffective for leachates with high organic contents but is beneficial for treating leachates from stabilised landfill sites, and for further "polishing" initially high-strength leachates following biological treatment (Chian and DeWalle, 1976). For a leachate with a high inorganic content, physico-chemical treatment, prior to subsequent biological treatment, minimises the possible effects of metal toxicity, corrosion and scaling (Scott, 1982). The efficacies of chemicals to remove colour, turbidity, heavy metals, calcium and magnesium have been well documented (Thornton and Blanc, 1973; Chian and DeWalle, 1976; Farooq and Velioglu, 1989; Swiderska-Bróz, 1991; Sletten et al., 1995). Each of these does, however, carry attendant costs.

The principal objective of this study was to determine a suitable, cost-effective treatment protocol for a high-strength leachate from a closed co-disposal landfill site.

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Materials and methods

Landfill leachate

Collected leachate from a closed co-disposal site in Gauteng was stored in 20 ℓ closed containers at 4°C until required. The site had been operated for 17 years before accepting domestic refuse only until the full capacity was reached. The range of products (8.1% w/w of the total waste) co-disposed at the site included pesticides, pharmaceutical and veterinary compounds, medical wastes, food processing wastes and phenolic wastes.

Phosphate supplement

For some aerobic studies KH_2PO_4 (0.38 g·ℓ^{-1}) and K_2HPO_4 (0.13 g·ℓ^{-1}) were added (1, Table 1).

Medium

The basic mineral salts medium described by Coutts et al. (1987) was used in the anaerobic studies (2, Table 1)

Batch cultures

Table 1 summarises the experimental details for the initial batch cultures. The leachate was diluted with glass-distilled water and the inoculum (15% w/v) was one-month-old refuse. For the aerobic cultures, 250 mℓ conical flasks, plugged with non-absorbent cotton wool, were used and were incubated (30°C) in

a New Brunswick Scientific rotary shaker (Model G-26) at 150 r·min^{-1}. The anaerobic cultures were made in 200 mℓ static screw-capped bottles equipped with hypodermic needles and syringes, for collection of the fermentation gases, and were overgassed with oxygen-free nitrogen (OFN). Sodium hydroxide (1M) was used to adjust the pH of selected cultures to 7. The total volume of each culture was 100 mℓ and incubation was at 30°C for 150 d in the dark. After incubation, the contents of the flasks and bottles were filtered through muslin cloth to remove the refuse before centrifugation at 8 000 r·min^{-1} x g for 20 min.

Microbial activity determination in batch cultures

A second set of aerobic batch cultures, with the leachate diluted with glass-distilled water to final leachate concentrations of 10%, 25%, 50%, 75% and 100% v/v, was made with microbial activity determined after 4 and 8 days by the fluorescein diacetate (FDA) bioassay.

FDA was dissolved in acetone (2 mg·mℓ$^{-1}$) and stored as a stock solution at 4°C. Potassium phosphate buffer (8.7g K_2HPO_4 and 1.3 g KH_2PO_4) was prepared and then diluted to one litre with distilled water and the pH adjusted to 7.6 with either NaOH (1M) or HCl (1M).

To flasks which contained 20 mℓ of leachate, 20 mℓ of phosphate buffer and 0.2 mℓ FDA were added. The flasks were incubated at 30°C in a New Brunswick Scientific rotary shaker (Model G-26) at 100 r·min^{-1} for 60 min. Each treatment was duplicated and a control, to which no FDA was added, was included. The FDA hydrolysis was terminated by adding acetone to a final concentration of 50% (v/v) (Schnürer and Rosswall, 1982). The flask contents were then filtered through No. 1 Whatman filter paper. The amount of

TABLE 1
DETAILS OF AEROBIC AND ANAEROBIC BATCH CULTURE CONDITIONS

Culture	Aeration		pH		Final leachate conc. (% of original)	Medium	Inoculum
	Aer.	Anaer.	Unadj.	pH7			
1	+		6.24		Undiluted	-	+
2	+		6.33		50%	-	+
3	+		6.94		10%	-	+
4	+		6.32	6.93	Undiluted	-	+
5	+		6.38	7.09	50%	-	+
6	+		6.86	7.01	10%	-	+
7	+		6.05		Undiluted	1	+
8	+		6.26		50%	1	+
9	+		6.69		10%	1	+
10	+		6.10	7.00	Undiluted	1	+
11	+		6.35	7.07	50%	1	+
12		+	6.78	6.99	10%	1	+
13		+	6.45		Undiluted	-	+
14		+	6.78		50%	-	+
15		+	7.08		10%	-	+
16		+	6.47	7.00	Undiluted	-	+
17		+	6.49	6.94	50%	-	+
18		+	6.92	7.00	10%	2	+
19		+	5.95		Undiluted	2	+
20		+	6.11		50%	2	+
21		+	6.35		10%	2	+
22		+	6.02	6.96	Undiluted	2	+
23		+	6.13	7.10	50%	2	+
24		+	6.20	7.20	10%	2	+
25	+		6.19		Undiluted	-	-
26	+		6.26		50%	-	-
27	+		6.45		10%	-	-
28	+		6.20	6.97	Undiluted	-	-
29	+		6.37	7.03	50%	-	-
30	+		6.47	6.95	10%	-	-
31	+		6.09		Undiluted	1	-
32	+		6.29		50%	1	-
33	+		6.48		10%	1	-
34	+		6.08	6.95	Undiluted	1	-

35	+		6.33	6.97	50%	1	-
36	+		6.80	6.95	10%	1	-
37		+	6.12		Undiluted	-	-
38		+	6.20		50%	-	-
39		+	6.38		10%	-	-
40		+	6.12	6.95	Undiluted	-	-
41		+	6.21	7.03	50%	-	-
42		+	6.44	7.01	10%	-	-
43		+	6.03		Undiluted	2	-
44		+	6.10		50%	2	-
45		+	6.32		10%	2	-
46		+	6.07	6.98	Undiluted	2	-
47		+	5.95	7.01	50%	2	-
48		+	6.40	6.99	10%	2	-

FDA hydrolysed was measured as absorbance at 490 nm with a Milton Roy Spectronic 301 spectrophotometer.

Continuous cultures

Two all-glass chemostats (Senior and Balba, 1984) (working volume 550 mℓ) were used. The aerobic chemostat was oxygenated with air (>v/v aeration) while the anaerobic chemostat was overgassed with OFN. A zinc acetate (0.1% w/v) trap was connected to the anaerobic chemostat to trap H_2S as zinc sulphide. The reservoir volumes were 2 ℓ and the influent diluted leachate (10% v/v) was introduced into each chemostat at a dilution rate (D) of 0.01 h^{-1}. Phosphate supplement was added to the aerobic chemostat influent reservoir to effect metal precipitation. Medium was added to the anaerobic chemostat influent reservoir. Gas samples for methane analysis were taken from the headspace of the chemostat. The chemostats were incubated at 25°C. After each full culture volume displacement, the effluent was recycled back through each chemostat.

Leachate physico-chemical treatment

The laboratory jar test of Thornton and Blanc (1973) was used. This involved adding lime (in increments to give final concentrations which ranged from 1 000 mg·ℓ^{-1} to 10 000 mg·ℓ^{-1}) to 500 mℓ of leachate with rapid mixing for 1 min. With low speed stirring, flocculation was allowed to proceed for 15 min before decanting into a graduated cylinder for measurement (after 1 h) of the settleable solids. The supernatant was then analysed.

Shortlands subsoil

The Shortlands subsoil (Table 2) used was collected from Ukulinga Farm, Pietermaritzburg. The soil was air-dried at ambient temperature and sieved (<2 mm) before use.

Leachate breakthrough curves

A perspex microcosm (height 12.5 cm, i.d. 5.5 cm) packed with the Shortlands subsoil and covered to exclude light was used to determine breakthrough curves at ambient temperature (+/- 21°C). Leachate was introduced at a rate of 0.5 mℓ·h^{-1} by a Type 202S Watson-Marlow peristaltic pump into the base of the column, and samples were collected hourly with a Gilson Model 203 microfraction collector for analysis. The microcosm study was terminated after 9 pore volumes changes.

Analytical methods

Volatile fatty acids and methane

Volatile fatty acids (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 OFN carrier gas was maintained at 30 mℓ·min^{-1}. For VFA analysis a stainless steel column (length 2 m, i.d. 4 mm) 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}. Acidified standards (500, 1 000 and 2 000 mg·ℓ^{-1}) were injected and the concentrations of VFAs calculated by peak area comparison. The standards and samples (1 $\mu\ell$) were acidified with formic acid (1% v/v). For methane analysis a glass column (length 1.45 m, i.d. 3 mm) packed with Poropak T (80/100

TABLE 2
SHORTLANDS SUBSOIL ANALYSIS

Soil properties	% (w/w)
Textural analysis	
- clay	60
- silt	34
- sand	6
Clay minerals	
- kaolinite	50
- chlorite	25
- interstratified	25
Organic carbon	1.85
Exchangeable cations	
- Na	0.29 $cmol_c·kg^{-1}$
- Ca	6.11 $cmol_c·kg^{-1}$
- Mg	5.41 $cmol_c·kg^{-1}$
- K	1.69 $cmol_c·kg^{-1}$
- Al	0.20 $cmol_c·kg^{-1}$
Cation exchange capacity (CEC)	13.70 $cmol_c·kg^{-1}$

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 comparing peak area response with those of standards prepared with pure methane (Fedgas).

Cations

Leachate cations were determined with a Varian Spectra AA-200 atomic absorption spectrophotometer. Standards were constituted with ultra-pure AAS reagents.

The conditions used for each metal analysis were as follows:

Ca^{2+}: Wavelength, 239.9 nm; Spectral Band Pass, 0.2 nm; Lamp current, 3 mA; Flame, air-acetylene.
Mg^{2+}: Wavelength, 202.5 nm; Spectral Band Pass, 1.0 nm; Lamp current, 3 mA; Flame, nitrous oxide-acetylene.
Na^{+}: Wavelength, 589.0 nm; Spectral Band Pass, 0.2 nm; Flame emission; Flame, air-acetylene.
K^{+}: Wavelength, 766.5 nm; Spectral Band Pass, 0.2 nm; Flame emission; Flame, air-acetylene.
Mn^{2+}: Wavelength, 279.5 nm; Spectral Band Pass, 0.2 nm; Lamp current, 5 mA; Flame, air-acetylene.
Zn^{2+}: Wavelength, 213.9 nm; Spectral Band Pass, 0.2 nm; Lamp current, 5 mA; Flame,air-acetylene.
Fe^{2+}: Wavelength, 248.3 nm; Spectral Band Pass, 0.2 nm; Lamp current, 5 mA; Flame, air-acetylene.

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 ($g \cdot \ell^{-1}$ 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 $\mu S \cdot cm^{-1}$, contained the following ($mg \cdot \ell^{-1}$ glass-distilled water): borate/gluconate concentrate, 20; acetonitrile, 120.

Samples (100 $\mu\ell$) were injected into an IC-Pak A column (4.6 x 50mm) which contained trimethylammonium functionalized polymethacrylate, water, lithium meta-borate and sodium gluconate (10 µm particle size). Standards of nitrite, nitrate, phosphate and sulphate (5, 5, 10, 5 $mg \cdot \ell^{-1}$, 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.

Chloride

The Mohr method was used (Basset et al., 1978).

Specific conductivity

Specific conductivity was measured with a Radiometer CDM83 conductivity meter.

pH

Culture supernatant pH was measured with a Crison MicropH2000 pH meter.

COD

Chemical oxygen demand (COD) was measured by the SA Breweries Method (Hoffman, 1986).

BOD_5

Biological oxygen demand (BOD_5) measurements were made by Umgeni Water Analytical Services Department.

Results and discussion

Leachate characterisation

The high-strength leachate used contained high concentrations of VFAs (Table 3) which suggested that the refuse mass was in the acidogenic phase of degradation.

The BOD_5 was, however, uncharacteristically low (8.8 $mg \cdot \ell^{-1}$) and the BOD:COD ratio suggested either a recalcitrant leachate (Ehrig, 1984) or one which contained bactericidal/ bacteriostatic components. Despite the low BOD:COD ratio, biological treatment was considered due to the high concentrations of labile VFAs.

The very high conductivity was also uncharacteristic of a leachate from a municipal refuse landfill and was more typical of a leachate from a hazardous waste site (Batstone et al., 1989). It has previously been reported that there may be little or no difference in the quality of leachate from a co-disposal site compared with municipal refuse leachate (Watson-Craik et al., 1992; Chu et al., 1994). Due to the high COD and low phosphorus concentration, the leachate was regarded as phosphate deficient.

TABLE 3
CHEMICAL COMPOSITION OF THE LEACHATE. WITH THE EXCEPTION OF pH, SETTLEABLE SOLIDS AND SPECIFIC CONDUCTIVITY, ALL RESULTS ARE EXPRESSED AS $mg \cdot \ell^{-1}$

Analysis	Concentration
pH	6.8-7.3
Settleable solids	$7 m\ell \cdot \ell^{-1}$
COD	30 000-53 000
BOD_5	8.8
VFAs	
- acetic	14 000
- propionic	2 600
- butyric	4 400
- valeric	2 600
- hexanoic	1 400
Spec. conductivity	51 900-52 400 $\mu S \cdot cm^{-1}$
Chloride	16 000
Sodium	5 700-14 700
Potassium	1 670-1 880
Sulphate	1 330-2 000
Phosphate	2-53
Ammoniacal-N	1 400
Nitrate	2-24
Nitrite	0
Magnesium	245-2 900
Calcium	1 300-3 400
Iron	145
Manganese	8.15-14.7
Zinc	2.0

Biological treatment

Batch cultures

Of the 48 batch cultures, treatments 1 to 24 were inoculated with refuse. All the cultures did, however, contain some (unquantified) inoculum via the leachate.

Despite possible loss of volatile fatty acids in the aerobic cultures by aeration, both the aerobic and anaerobic batch cultures effected no significant COD reductions of the undiluted and 50% (v/v) diluted leachate. Reductions in the COD were only obtained with a final leachate concentration of 10% (v/v) (Fig. 1) which implicated the presence of bactericidal/bacteriostatic components in the original leachate and coincided with the low BOD_5 value recorded.

For the aerobic cultures, the highest COD reduction (53%) was obtained with phosphates added but no pH adjustment (Treatment 33). With the anaerobic cultures, Treatment 45, with nutrients added but no inoculum, effected a 54% reduction in COD while pH adjustment to 7 (Treatment 48) further promoted the COD reduction. Methane was detected in this treatment only.

Determination of microbial activity, as measured by FDA, in the batch cultures of different leachate dilutions showed that after 4 d significant activity was only apparent when the final leachate concentration was 25% (v/v) of the original (Fig. 2). After 8 d, low absorbance readings were obtained for the higher (50% (v/v)) leachate concentrations which confirmed limited microbial activity.

Continuous cultures

In continuous culture (with recycle) the anaerobic treatment of diluted (10% v/v) leachate proved relatively ineffective even in the presence of added mineral salts (Fig. 3). Hydrogen sulphide was evolved but no methane, probably due to the kinetic advantage of sulphate-reducing bacteria (Widdel, 1988) compared with methanogens (Parkin et al., 1990). Also, the evolved H_2S could have inhibited the methanogens (Khan and Trottier, 1978).

A significantly higher COD reduction (76.6%) characterised the aerobic chemostat treatment after two culture volume displacements (Fig. 3). Further effluent recycling did not, however, effect further COD reductions, even after resupplementation with phosphates. A retention time >8 d was, therefore, necessary for effective COD removal. This time was somewhat longer than the 5 d reported by other workers (Scott, 1982; Robinson and Maris, 1983).

Following the preliminary chemostat study, two aerobic continuous cultures (10% and 25% v/v final leachate concentration) were established. Tables 4 and 5 show the results obtained after two culture volume displacements.

For both chemostats, COD reductions >72% were recorded. These were attributed to labile VFA removal as exemplified by pH increases to >8.5. No ammoniacal-N removal was apparent, probably due to the high sensitivity of the nitrifying bacteria to various metals and organic compounds (Blum and Speece, 1991).

Physico-chemical treatment

pH increases during biological treatment, combined with aeration through stirring, promoted precipitation of heavy metals from the leachate. Addition of phosphate also precipitated the metals as insoluble orthophosphates. Although precipitation of inorganic solids does not affect biological processes it can cause opera-

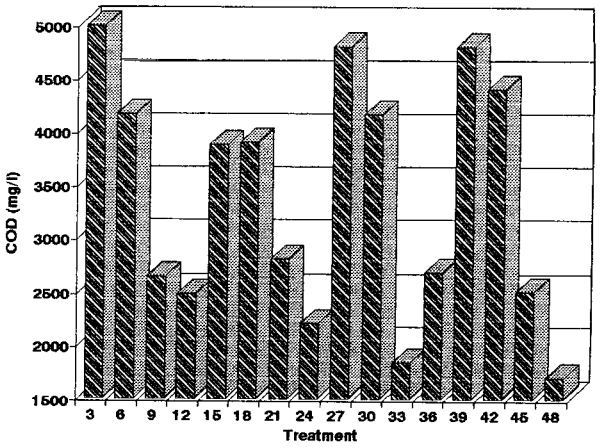

Figure 1
Residual CODs of aerobic and anaerobic batch cultures of 10% (v/v) leachate after 150 d incubation at 30°C. For details see Table 1

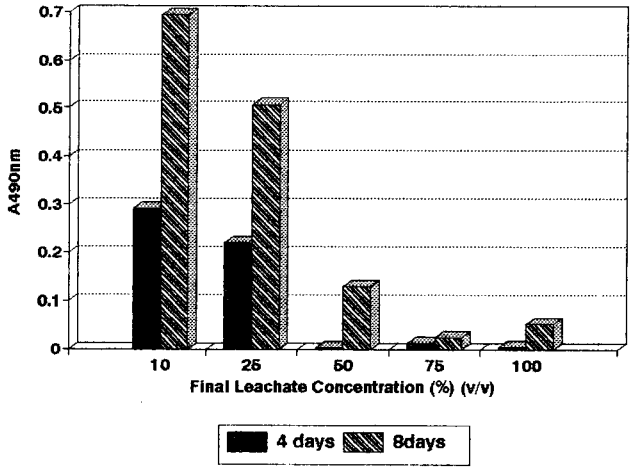

Figure 2
Microbial activity, as measured by fluorescein absorbance, in batch cultures of different leachate concentrations

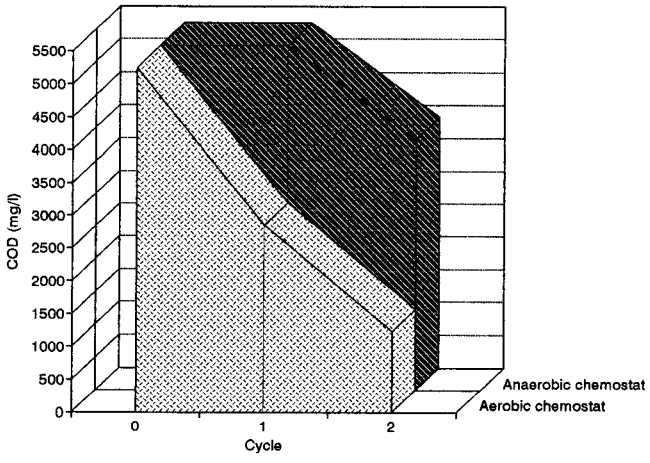

Figure 3
COD values of the aerobic and anaerobic chemostat effluents during two full culture volume displacements

TABLE 4
LEACHATE ANALYSES BEFORE AND AFTER PHOSPHATE SUPPLEMENTATION AND TWO CULTURE VOLUME DISPLACEMENTS IN AN AEROBIC CHEMOSTAT OPERATED WITH A DILUTED (10% V/V) LEACHATE

Parameter	Pre-supplementation concentration ($mg \cdot \ell^{-1}$)	Post-supplementation concentration ($mg \cdot \ell^{-1}$)
COD	3500	954
pH	6.5	8.6
VFAs	2531	0
Ammoniacal-N	180	190
Nitrite	0	0
Nitrate	0	0
Phosphate	0.2	32.3
Sulphate	200	224

TABLE 5
LEACHATE ANALYSES BEFORE AND AFTER PHOSPHATE SUPPLEMENTATION AND TWO CULTURE VOLUME DISPLACEMENTS IN AN AEROBIC CHEMOSTAT OPERATED WITH A DILUTED (25% V/V) LEACHATE

Parameter	Pre-supplementation concentration ($mg \cdot \ell^{-1}$)	Post-supplementation concentration ($mg \cdot \ell^{-1}$)
COD	8750	2140
pH	6.8	9.2
VFAs	6247	0
Ammoniacal-N	450	460
Nitrite	0	0
Nitrate	0	0
Phosphate	0.5	4.6
Sulphate	489	502.6

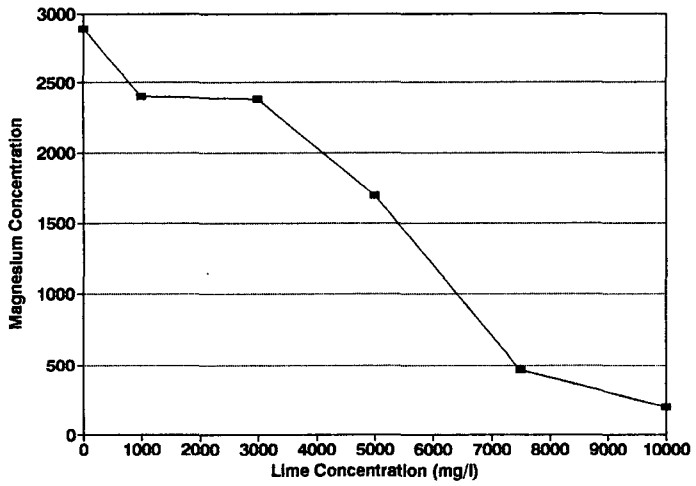

Figure 4
Leachate supernatant residual magnesium concentrations in response to lime additions

tional problems (Ehrig, 1984) such as scaling and clogging (Knox, 1985).

To identify a cost-effective physico-chemical treatment to effectively reduce the inorganic content of the leachate prior to biological treatment, lime (as $Ca(OH)_2$) and ion exchange with Shortlands subsoil were considered. Lime is the most conventional coagulant used in waste-water treatment due to its low cost and availability (Ho et al., 1974). The use of soil as an adsorbent is a cost-effective ion exchange treatment option, particularly if the soil is present in the vicinity of the landfill.

Lime

A visible reduction in leachate colour intensity (dark brown to a clear yellow) followed the addition of 5 000 $mg \cdot \ell^{-1}$ lime. The colour removal was attributed to the precipitation of insoluble ferric hydroxide ($Fe(OH)_3$) colloids. Subsequent chemical analysis confirmed that the iron concentration was negligible.

High magnesium concentrations (Table 3) were similarly lowered once the pH was increased to 9.5 (3 000 $mg \cdot \ell^{-1}$ lime). The magnesium concentrations in the supernatants of the different treatments are shown in Fig. 4. Zinc and manganese, which were present in low concentrations in the untreated leachate (Table 3), were removed by addition of 1 000 $mg \cdot \ell^{-1}$ lime.

Despite these encouraging results, problems arose from lime treatment. There was a progressive, but unacceptable, increase in the calcium concentration of the supernatant which could effect scaling and corrosion. Another drawback was the high pH which resulted in the displacement of ammonia which is an atmospheric pollutant (Harrington and Maris, 1986).

Ion exchange

Adsorption and desorption are the two major abiotic processes which affect contaminant transformations in soils. A high cation exchange capacity and good hydraulic conductivity are prerequisites for a soil to function as an effective ion-exchange system. Adsorption of ions on the cation exchange complex depends on the valency, hydrated form diameter and the type and concentration of other ions present in the soil solution (Alloway and Ayres, 1993).

As with lime treatment, soil filtration/biofiltration resulted in a leachate colour reduction from dark brown to clear yellow in all samples probably through removal of ferric hydroxide colloids. The quantities of metals adsorbed by the soil were determined by plotting C/C_o (effluent metal concentration/ influent metal concentration) against pore volume (Fig. 5 A to F). When $C/C_o = 1.0$, the microcosm was considered to be in a steady state, with subsequent adsorption minimal (Knox et al., 1993). The results showed that CEC adsorption of iron(III) was favoured, probably since the highly charged cation competed well for clay exchange sites occupied by divalent or monovalent ions. Although zinc should have been preferentially adsorbed, its breakthrough curve indicated a degree of desorption probably due to the mass action effect of a higher concentration of an individual ion, such as the

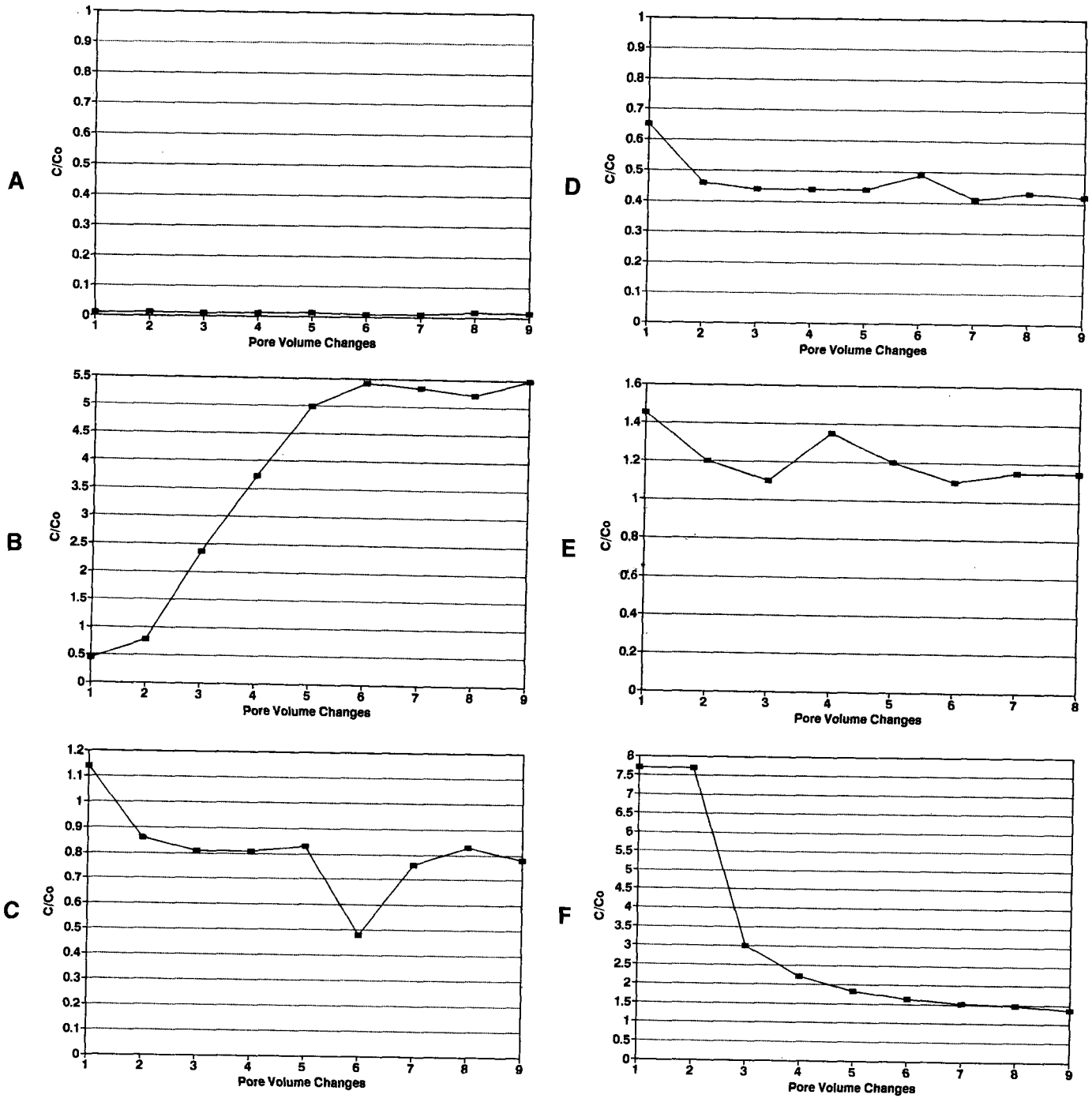

Figure 5

Changes in iron (A), potassium (B), calcium (C), magnesium (D), zinc (E) and manganese (F) adsorptions to Shortlands subsoil in relation to pore volume changes

magnesium, calcium or sodium, with less adsorption power (Knox et al., 1993). The microcosm effluent potassium concentrations progressively increased throughout the study following, possible, exchange with iron(III), magnesium or calcium ions. When a cation is removed from a wastewater by ion exchange, there is, generally, a subsequent release of some other cation from the soil. The manganese concentrations eluted from the microcosm were higher than the influent concentration probably due to the reductive dissolution of manganese oxides, which are ubiquitous in soils, by substituted phenols (Stone, 1987) or other organic compounds (Stone and Morgan, 1984) present in the leachate.

Conclusions

Due to the high COD and volatile fatty acid concentrations, biological treatment was considered for the leachate. Use of anaerobic treatment is regarded by many as the preferred option particularly for high-strength organic waste waters (Lin, 1991). In our study aerobic treatment was, however, identified as the better option. Aerobic biological treatment effected a COD reduction of 74% when the influent leachate concentration was <25% (v/v). Dilution may not be a cost-effective option, however, unless treated leachate is used as the diluent.

Automated aerated lagoons are well established in the U.K. (Robinson et al., 1992) as cost-effective, simple treatments. The higher ambient temperature of South Africa should favour this technology provided that oxygen is not limiting.

Aerobic treatment does, however, have problems such as production of large volumes of sludge. Failures of full-scale aerobic leachate treatment plants due to hydraulic and/or organic overloading, phosphate limitation and inadequate aeration have been reported (Harrington and Maris, 1986).

To minimise the inorganic content of the leachate and, thus, facilitate improved biological treatment, pretreatments with lime or by ion exchange were considered. Unfortunately, for this particular leachate, although iron and colour were removed, the concentrations of magnesium and calcium were not effectively reduced. The dissolution of manganese oxide in the Shortlands subsoil, due to the presence of leachate organic matter, also limited its use as an ion-exchange medium. Further studies of ion exchange, which focus on higher CEC materials, such as activated carbon, should, therefore, be made.

Acknowledgements

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