Effects of sewage sludge and refuse composition on the rate of degradation and leachate quality of co-disposed waste in a water-deficient environment

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Abstract

At the Goudkoppies landfill to the south-west of Johannesburg co-disposal is part of the landfill operation. Laboratory and field tests were conducted to model the landfill processes and the effects of sewage sludge addition. To test the hypothesis that the availability of nitrogen, phosphorus and moisture in the refuse and sewage sludge determine the effects of co-disposal, 19 lysimeters were monitored for 534 days. A factorial design approach was used to choose the lysimeter contents to simulate local landfill conditions. Degradation rates and ammonia concentrations in the leachate were found to depend on the interactions between the nutrients and moisture, rather than on any individual variable. Co-disposal led to a decrease in COD and nickel concentrations, while levels of ammonia and phosphorus in the leachate were elevated in comparison to the refuse-only test. From field tests it was concluded that co-disposal can be practised in the Highveld region without the formation of leachate, while enhancing the rate of refuse stabilisation.

Introduction

In South Africa urban areas are growing rapidly, placing a burden on existing infrastructure. To ensure that cities can accommodate the growing population, there is a need for the rapid provision of township services, most significantly, water and sanitation.

The importance of solid waste management under these circumstances is often neglected or unde-restimated. Solid waste has both direct and indirect health and environmental implications with uncontrolled refuse attracting vermin and other disease vectors, causing an odour nuisance and washing into streams, thus threatening the supply of potable water. Waste water is another form of waste produced by urbanised communities and is a further health risk if not properly managed. Although waste-water treatment plants are able to purify water and allow for its return to streams, a by-product of the process, sewage sludge, remains. Cost-effective, hygienic and environmentally sound disposal is required to deal with the sludge because of its high organic content, heavy metal concentrations and the presence of pathogens.

Recent years have seen the development of sanitary landfills where refuse is deposited and left to decompose under controlled conditions. Moisture entering the landfill or released during the decomposition of waste percolates through the refuse, carrying pollutants in solution and forms leachate. To reduce the risk of ground-water pollution by leachate, sanitary landfills are lined with clay and/or a geomembrane. Soil is used to cover the refuse at the end of each day and this excludes vermin and reduces odour generally associated with decomposing waste.

A method of dealing with sewage sludge in other countries has been to mix it with refuse on landfills in a process called co-disposal (Garvey et al., 1993, amongst others). This is considered relatively cheap, compared to other methods of sewage sludge disposal, as well as being hygienic and environmentally acceptable. Possible benefits to the landfill degradation processes could also be derived (e.g. Buivid et al., 1981).

Landfill degradation processes

The quality and volume of the landfill degradation products, leachate and gas, are determined by the dominant process at any particular time. As the decomposition progresses, various bacterial populations dominate, each releasing differing metabolic products into the landfill. Some of these products are transported from the landfill by leachate and gas.

Initially, oxygen trapped during the placement and compaction of the waste is used by aerobic organisms to break down organic matter for energy, releasing carbon dioxide. There is a shift to fermentative or anaerobic processes as the oxygen is depleted. Leachate pH declines as fatty acids and alcohols accumulate, and due to the acidity, metals are mobilised by the leachate. At this time, carbon dioxide concentrations reach a peak of up to 90% by volume (Stegmann and Spendlin, 1987). During a third phase the simple acids and alcohols formed in the previous phase are used by methane-producing bacteria. Metal concentrations, chemical oxygen demand (COD) and acidity of the leachate reduce and a relatively constant methane concentration of 45 to 55% by volume is attained and gas production rate reaches its peak. When the methane and carbon dioxide proportions in the gas are almost equal and remain constant, a fourth, stable methanogenic phase has been reached. Gas production slows as the carbon source is depleted or nutrients become limiting but may continue for many years.

Because it is a biological process, the rate at which degradation occurs is strongly linked to the conditions in the landfill. During each of the phases described above, different groups of microbes dominate, depending on the environment within the waste. Aerobic, facultative or anaerobic organisms are active at different times as the concentrations of oxygen in the landfill change. Differing conditions may exist simultaneously at different locations in a landfill. Acid-producing bacteria are suited to low pH conditions and are the dominant group during the transi-

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Received 28 August 1997; accepted in revised form 8 December 1997

tion or acid-producing phase. Methanogenic bacteria, although also anaerobic, require more alkaline conditions and use simple acids as a substrate while releasing methane. As methanogenic bacteria use up acids, more are produced by the acid-producing microbial groups. If acids are formed faster than they can be consumed, there is an accumulation of these products which makes conditions unfavourable for the methanogens.

Co-disposal theory

Generally, landfill research has focused on enhancing the rate of degradation (Stegmann and Spendlin, 1987; Kinman et al., 1987, amongst others). Potential benefits that are associated with an increased rate of decomposition include:

- A shortened acid-producing phase before methanogenesis. This reduces the period during which acidic leachate containing high COD, metal and acid concentrations is formed. Concentrations of parameters such as COD and iron reduce with an increase in leachate pH (COD reduces as a result of acid utilisation by methanogens). By reducing the duration of the acid phase, the risk of groundwater contamination is lowered and ultimate leachate treatment costs are reduced.
- Gas production is concentrated over a shorter period. The total volume of gas production remains unaltered, but by shortening the period over which the peak in the gas release occurs, the extraction of methane is economically more viable. Landfill sites can be reused earlier, since the gas potential is exhausted sooner than at slowly decomposing landfill sites.
- Settlement of the waste bulk occurs sooner than at landfills where degradation is slower. A greater volume of waste can thus ultimately be contained at the landfill site.

Literature on sewage sludge co-disposal does not give a clear indication of the effects that sewage sludge has on the landfill degradation processes, leachate and gas quality and volume. Barlaz et al. (1987) and Chapman and Ekama (1991) (and others) found that methane production is inhibited and the time taken for the waste to reach stable methanogenic conditions is extended when sewage sludge is added. Contradictory conclusions have been proposed by others (e.g. Blakey, 1991; Stamm and Walsh, 1988) who argue that earlier methanogenic conditions are experienced in co-disposal studies than in refuse-only controls.

Conflicting data with respect to leachate quality are also discussed in various studies. For example, Blakey (1991) showed that the cumulative mass of constituents leached in co-disposal tests was less but the concentrations were higher than in refuse-only degradation experiments.

There appear to be two main reasons for the contradictions: varying proportions of sludge and refuse are combined, and different types of sewage sludge are mixed with non-homogeneous refuse. In particular, the volume of sludge and the moisture content of the sludge are the main variables. Conditions under which the experiments were carried out were also particular to each study, e.g. some tests were performed outdoors, subject to seasonal temperature variations whilst others were conducted in controlled, heated environments. Comparing the results from various studies described in the literature, it becomes clear that conclusions are specific to each test series and that no general trends can be identified.

The aim of the research is to assess the effects of sewage sludge co-disposal on landfill degradation rates and leachate quality. Contradictions in the literature have led to the hypothesis that various components of the refuse and sewage sludge, namely moisture content and the availability of nutrients, govern decomposition rates. Further, these parameters were considered to also have interactive effects. The extent of interactive and individual effects were studied under laboratory conditions to test this hypothesis.

Factors affecting sewage sludge and refuse decomposition

Biological processes, although poorly understood, can be influenced to a greater extent than the chemical or physical decay mechanisms through controlling the landfill environment. The rate of degradation depends on a complex interaction of factors. Availability of substrate, nutrients and moisture are essential for biological activity and by adjusting the relative availability of these parameters, as well as pH and the presence of toxins, the environment is altered to suit selective populations of microbes and thereby, the products of their metabolism.

Importance of water

Organic matter contains a high proportion of water. For example, bacterial cells are composed of 75 to 90% water by mass. In order for microbes to produce cells and grow, sufficient moisture must be present in the landfill environment.

Water is the medium for the transfer of substrate and nutrients within the landfill. Moisture is also required for the dilution of toxins to levels that are not harmful to organisms active in the waste. Low moisture availability can inhibit growth and the rate of degradation in the landfill is enhanced as the water content of the waste is increased (Rees, 1980), to a limit above which there is no further improvement.

Areas where annual evapotranspiration exceeds precipitation are classified as water deficient (Ball et al., 1993) and as a result of the dry conditions, degradation of waste in the landfill is likely to be very slow. Not only is the annual precipitation lower than potential evapotranspiration in the Highveld region, but landfill operations are designed to limit water entering the waste. Runoff is high and drainage from surrounding areas is diverted from the landfill. Daily cover retards infiltration of precipitation although some infiltration may occur while the waste is still uncovered. According to Ham and Barlaz (1987), very little methane is detected at landfills in dry climates. However, in gas sampled at field tests at the Goudkoppies landfill in Johannesburg, high concentrations of methane were detected (Röhrs et al., 1995).

Sewage sludge has a high moisture content (for example, sewage sludge from the Goudkoppies Waste Water Treatment Works has an average water content of 400% by mass). Water is present as free or organically bound moisture. By controlling the volumes of sludge used in co-disposal, extra moisture could be incorporated into the landfill without exceeding the field capacity i.e. leachate will not form, even with the increased moisture in the landfill.

The role of nutrients

The most important nutrients required for growth of microbes are nitrogen and phosphorus. Nitrogen is the essential element in protein, while phosphorus is used for energy storage and transfer in the form of ATP (adenosine triphosphate). It has been suggested that refuse is nutrient-deficient. If this is the case, nutrients are rate-limiting in the degradation of organic compounds in the landfill. When nutrients are not available in the correct proportions, the number of organisms that can be sustained is based on the availability of the nutrient of which there is the lower supply. An optimal ratio of C:N:P of 100:5:1 was proposed by De Baere and Verstraete (1984). The actual relationship of these elements in the refuse used in their research was 100:1.3:0.2.

The hypothesis of a nutrient deficiency in refuse is supported by Pacey (1989) and Cossu et al. (1987). Biochemical methane potential tests (BMP) were conducted on refuse samples obtained from large-scale experiments where various techniques for the enhancement of degradation had been tested. To test for a phosphate deficiency, Pacey (1989) compared the results from a reactor containing sufficient phosphate to those where no phosphate had been added. Refuse from three of the four experiments was found to be limited by the availability of phosphorus since methane production measurements in the BMP analyses were significantly reduced after 133 d of incubation when compared to the test where phosphorus had been supplemented (Pacey, 1989). Nitrogen was also likely to have been a limiting nutrient (Pacey 1989). Cossu et al. (1987) found that degradation was enhanced by nitrogen addition and even more when refuse was supplemented with phosphorus. Rees (1980), however, was of the opinion that nutrients are not limiting in landfills since phosphorus and nitrogenous compounds appear in leachate and thus it is argued that refuse contains nitrogen and phosphorus in excess of the amounts forming bacterial biomass. However, refuse is not homogeneous and degradation occurs at differing rates throughout the landfill; nutrients are not evenly distributed and are used at different rates; areas of the landfill have differing field capacities, allowing leachate to remove nutrients even if they are not present in excess quantities.

Sewage sludge is very difficult to process because of the relatively high nitrogen, phosphorus and heavy metal concentrations (Ratsak et al., 1996). For South African sewage sludge the median nitrogen and phosphorus concentrations are 2.89% (dry mass) and 1.35%, respectively (Smith and Vasiloudis, 1989). Since refuse is considered nutrient-deficient, and sewage sludge has an excess of these nutrients, it can be assumed that the effect of co-disposal on the rate of degradation is strongly related to the nutrient availability. Components of the refuse and sludge may be complementary in the processes of decomposition.

Sludge-to-refuse ratio

Degradation is the result of complex interactions of groups of microbes with conditions in the landfill determining the size of the various populations. For example, bacteria active in the early stages of decomposition break down complex organic matter to simple acids. These bacteria can tolerate low pH environments, whereas the methane-producing bacteria require neutral or alkaline conditions.

Digested sewage sludge contains anaerobic bacteria which are able to degrade organic matter in the sewage sludge and produce methane. These bacterial populations can form an inoculum to the landfill. Since these microbes were active in the anaerobic digesters at the sewage treatment works, they have already been subjected to high heavy metal concentrations, unlike bacteria in the landfill. Bacteria in a landfill where co-disposal has not been practised may be inhibited by a shock load of organic material. However, a recent study to enumerate anaerobic microbes on fresh refuse found that most of the trophic groups required for degradation are already present on the waste brought to the landfill (Qian and Barlaz, 1996).

Digested sewage sludge is neutral or slightly alkaline and it is not known whether the buffering capacity is sufficient to maintain conditions suited to methanogenic bacteria. Barlaz et al. (1987) found that the addition of sewage sludge to refuse caused a build-up of carboxylic acid, and a concomitant reduction in pH.

Voids remain in the landfill, even when the refuse is well compacted. Sewage sludge co-disposal does not add significant extra volume to the landfill since it fills the voids in the refuse. An increased density can thus be achieved through the addition of sewage sludge. For example, at Coastal Park landfill an 8% increase in compaction density was recorded at co-disposal test sections using sewage sludge with 2 to 3% solids (Novella et al. 1994).

By altering the load of sewage sludge on the landfill, the proportions of nutrients, moisture and other components of the refuse and sludge are varied. The effect that sewage sludge has on the landfill degradation rate and leachate and gas quality and volume can be changed by altering the refuse-to-sludge ratio.

Experimental programme

Degradation rates of refuse are related to the composition of the waste. For example, Wolffson (1985) conducted experiments on different fractions of refuse and found that paper and grass produced the same cumulative volume of methane, but the grass took 40 d longer to attain methanogenic conditions. Changing the refuse composition from one experiment to another is therefore likely to impact on the rate at which methanogenesis is attained and the quality of the gas and leachate produced.

Added to the variable refuse composition is the variability of sewage sludge from different sources. Many waste-water purification and sterilisation methods exist and they give rise to sludge with different characteristics e.g. limed or anaerobically digested sludge. This alters the availability of substrate, nutrients, moisture and acidity levels in the sludge which impacts on the ease with which the organic material can be degraded.

Many of the research programmes described in the literature test the effect of a single variable e.g. the effect of moisture on the methane production rate (e.g. Rees, 1980). Because the waste is of a variable nature, it is likely that the measured response cannot be entirely attributed to a single parameter, but to combinations and interactions which occur. Testing only one factor at a time, the implicit assumption is made that if the level of any of the other parameters in the experiment were altered, the results would remain unchanged.

Factorial design

Problems inherent in the conventional experimental methods can be addressed using factorial design. Experiments are set up to assess the effects of individual variables, as well as their interactions. Replication is inherent in the system, and the number of tests does not need to be increased to achieve improved confidence in the data (For more detail, refer to Box et al., 1978).

Parameters are tested at different levels or concentrations, so that the effects of these variables can be determined. Any number of levels can be chosen for each variable and they need not all have the same number of levels.

The importance of an energy source and the availability of nutrients and moisture during biodegradation has already been discussed. Although these are not the only parameters affecting the complex landfill processes, they are factors which can be altered by changing the relative proportions of refuse and sewage sludge in the landfill. Processes in the landfill are influenced by a multitude of factors, many of which may be interrelated. Because of these interactions, it is not sufficient to test the effects of parameters such as moisture and nutrient availability in isolation. By varying the main constituents of sewage sludge and refuse, the effects of these parameters on the landfill processes and products may be observed.

The components of the refuse and sewage sludge, discussed above, which were likely to have the largest impact on the rate of degradation and the quality of gas and leachate were used in the factorial design. To test the hypothesis that the components of the refuse and sludge determine the outcome of co-disposal experiments, moisture, nitrogen, phosphorus and sludge-to-refuse ratio were used in the experimental programme. The effects of these parameters and their possible interactive effects on the landfill processes were investigated in a series of 16 lysimeters. Each of the variables was tested at two levels and all combinations of these were considered. Two further lysimeters were filled with a mixture in which the parameters of interest were mid-way between the high and low values used in the factorial design, while the last lysimeter contained only refuse (The lysimeter contents are given in Table 1).

Waste source

Refuse for the experiment was obtained from the Goudkoppies landfill site situated in the south-west of Johannesburg. Because this was the only landfill in the area where co-disposal occurs, it was used as a case study to describe changes to degradation processes occurring as a result of sewage sludge inclusion in the waste.

Goudkoppies landfill receives refuse from Soweto, Diepmeadow, Ennerdale and the south-western suburbs of Johannesburg. Refuse arriving at the site has a varied composition which depends on the area of origin (Röhrs et al., 1995). Waste generated in Johannesburg has far higher levels of packaging materials than refuse from Soweto where the waste contains a high proportion of ash. Goudkoppies landfill site has a different refuse composition to other landfill sites and co-disposal results from First-World countries are unlikely to apply (Refer to Table 2 for chemical composition of refuse used in the experiments. Although only nickel concentrations in the leachate are discussed later, a zinc equivalent is included as an indicator of metal content of the samples).

Sewage sludge co-disposal has been part of the Goudkoppies landfill operations since its opening in 1990. Quarterly groundwater tests have shown that the water quality has not changed since the landfill was opened (Henning, 1993). It was assumed that no leachate had formed because the groundwater was uncontaminated and, based on a water balance, no leachate was expected. No test had, however, been conducted on the rate of degradation or gas composition prior to the present study.

Anaerobically digested sewage sludge from the waste-water treatment plant adjacent to the landfill has been used in the co-disposal operations at the Goudkoppies landfill site. Sludge for the lysimeter and field tests was therefore obtained from the same source (Röhrs, 1995). (The sludge and refuse compositions are presented in Table 2).

TABLE 1
INITIAL LYSIMETER CONTENTS

No.	Sludge:MSW (wet weight)			Moisture	Nitrogen		Phosphorus		
	Ratio	MSW (kg)	Sludge (kg)	(% solids)	(g/kg DS)	Added 1	(g/kg DS)	Added 2	Added 3
1	1:4	64.0	16.0	19.0	50.53	91.42	36.43	10.09	38.66
2	1:4	64.0	16.0	19.0	50.53	91.42	33.00		
3	1:4	64.0	16.0	19.0	40.00		36.43	10.09	38.66
4	1:4	64.0	16.0	19.0	40.00		33.00		
5	1:4	64.0	16.0	84.5	52.53	406.56	40.43	44.88	171.93
6	1:4	64.0	16.0	84.5	52.53	406.56	37.00		
7	1:4	64.0	16.0	84.5	42.00		40.43	44.88	171.93
8	1:4	64.0	16.0	84.5	42.00		37.00		
9	1:12	73.85	6.15	19.0	50.53	35.14	36.43	3.88	14.86
10	1:12	73.85	6.15	19.0	50.53	35.14	33.00		
11	1:12	56.53	4.71	19.0	40.00		36.43	2.97	11.38
12	1:12	73.85	6.15	19.0	40.00		33.00		
13	1:12	73.85	6.15	84.5	52.53	156.28	40.43	17.26	66.09
14	1:12	73.85	6.15	84.5	52.53	156.28	37.00		
15	1:12	73.85	6.15	84.5	42.00		40.43	17.26	66.09
16	1:12	73.85	6.15	84.5	42.00		37.00		
17	1:6	68.57	11.43	51.7	49.85	138.85	38.90	15.33	58.70
18	1:6	68.57	11.43	51.7	49.85	138.85	38.90	15.33	58.70
19	-	80	0	-	-	-	-	-	-

1 Ammonium nitrate
2 Sodium dihydrogen orthophosphate
3 Disodium hydrogen orthophosphate anhydrous

TABLE 2 CHEMICAL ANALYSIS OF REFUSE AND SEWAGE SLUDGE				
Analysis	**Units dry mass**	**Sludge 1 ***	**Sludge 2 ****	**Refuse**
% solids		19	84.5	76
Volatile matter	g/kg	622	649	346
Kjeldahl nitrogen	g/kg	40	42	7
Total phosphorus	g/kg	33	37	2
Nickel	mg/kg	250	450	35
Specific energy	MJ/kg	12	13	8
Zinc equivalent	mg/kg	4760	6880	720

* Belt-pressed sludge
** Sun-dried sludge

Nitrogen concentrations in South African sewage sludge range from 17 to 50 gN/kg dry solids (DS), whereas lower concentrations over a smaller range are typical in the United Kingdom (15 to 25 gN/kg DS) (Smith and Vasiloudis, 1989). The average nitrogen content of the sludge from the Goudkoppies Waste-Water Treatment Works is very high, 45 gN/kg DS (Goudkoppies Works annual averages) whereas the median for South African sewage sludge is 28.9 gN/kg DS (Smith and Vasiloudis, 1989).

Concentrations of phosphorus in the sludge are near the upper limit found in South Africa - 4 to 41 gP/kg DS (Smith and Vasiloudis, 1989). The 30 gP/kg DS concentration of phosphorus is almost double the highest values (18 gP/kg DS) (Smith and Vasiloudis, 1989) measured in the UK.

Experimental procedure

Two types of anaerobically digested sewage sludge obtained from the Goudkoppies Waste-Water Treatment Works were used: sun-dried sludge (18% gravimetric water content) and sewage sludge directly from the belt-press (426% gravimetric water content). Samples of refuse were chosen randomly from the Goudkoppies landfill site and subsequently shredded and mixed to ensure relative homogeneity. The results of the chemical analyses conducted on the refuse and sewage sludge are presented in Table 2.

Gas-tight, 220 ℓ plastic drums were used for the lysimeters. (Refer to Fig. 1). Fittings with gate valves were added as irrigation, leachate and gas sampling ports. A radial arrangement of perforated tubes on the underside of the lid provided an even distribution of water onto the surface of the waste mixture. Leachate was directed to the outlet by a sloping, geomembrane-covered base overlain by a layer of 19 mm crushed stone (Röhrs, 1995).

Each lysimeter contained 80 kg of co-disposal mixture where the mass of sludge and refuse was based on the proportions needed to implement the factorial design. Chemicals required to supplement the nitrogen and phosphorus levels of the sludge for the experiment were estimated from the average concentrations recorded over the preceding year at the Goudkoppies Waste-Water Treatment Works. Following the chemical analyses of the sludge, slight additions had to be made to the chemicals in some lysimeters. These additional chemicals were introduced into the lysimeters with the irrigation water 20 d after the experiments had been sealed.

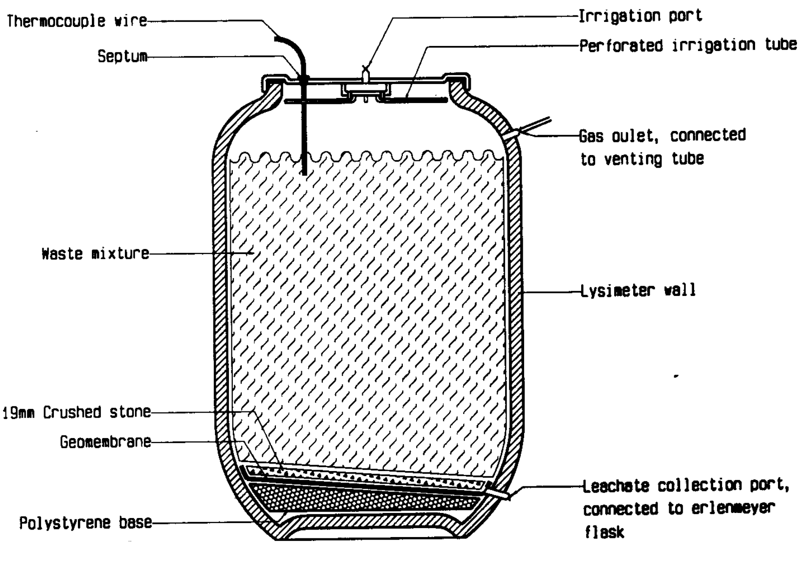


Figure 1
Schematic cross-section through lysimeter (Not to scale)

Urea has been used by some researchers to supplement nitrogen levels when remedying nutrient deficiencies (Leuschner, 1987, Britz and Van der Merwe, 1993). Since it is a product of protein degradation, urea would increase the nitrogen concentrations in the form of a compound already present in the landfill. However, for bacteria to use urea ($CO(NH_2)_2$) as a nitrogen source, it is split into carbon dioxide and ammonia (Atlas, 1984). The gas composition is one of the parameters conventionally used to monitor waste degradation in lysimeters and changes to the carbon dioxide levels would affect the interpretation of the data.

A non-selective growth medium that does not favour the establishment of any particular species of microbe was described by Atlas (1984) and contained, among others, ammonium nitrate and a phosphate buffer. Nitrogen levels were therefore increased using ammonium nitrate. Sodium dihydrogen orthophosphate and disodium hydrogen orthophosphate were used in the proportions required to produce an 0.02 M buffer, except that water was not added.

Two levels of moisture were attained by using sludge obtained directly from the belt presses at the Goudkoppies Waste-Water Treatment Works and sludge which had been sun-dried.

To reduce some of the complexity of modelling landfills at laboratory scale, the following parameters were controlled:

- Moisture: Lysimeters were irrigated with 0.5 ℓ rain water per week. The value represents the estimated excess of precipitation over evaporation during the summer rainfall months at the Goudkoppies landfill.
- Temperature: All lysimeters were maintained at 30°C, which is considered optimal for degradation (Cecchi et al., 1992; Blakey, 1991). Heater tapes controlled by thermostats, together with a fibreglass insulation layer, were used to maintain the required temperature.
- Refuse homogeneity: Refuse was shredded to reduce the size to something more appropriate to the scale of the experiments and enable a more homogeneous and representative sample to be placed in each lysimeter.

Results and interpretation

Time taken to reach methanogenic conditions

Changes in landfill processes occur gradually and a convenient parameter was required to assess the rate at which degradation was occurring in each of the lysimeters (See Table 3 for the duration of the lag phase in each lysimeter). For the purpose of the data analysis, the time taken for the lysimeter contents to reach methanogenic conditions is defined as the time when the first increase in leachate pH followed by a sharp upward trend was observed. It is used to simplify comparative analyses; any of a variety of indicators could have been used. COD, sulphate and iron concentrations, for example, are also linked to the onset of methanogenesis and show a sharp decline at the end of the lag phase. However, graphs of these concentrations have more fluctuations than the relatively smooth pH curves. Further, concentrations of metals depend on the acidity of the leachate.

Based on the change in the leachate pH, the time taken for co-disposal lysimeters to reach methanogenic conditions is less than for the refuse-only test (Students' t test, $\alpha = 0.001$). This implies that the rate at which degradation occurred in the lysimeters was enhanced by the addition of anaerobically digested sewage sludge, both belt-pressed and sun-dried sludge.

TABLE 3
TIME SINCE SEALING OF DRUMS FOR WASTE TO REACH METHANOGENESIS

Lysimeter	Time to methanogenesis (d)	Lysimeter	Time to methanogenesis (d)
1	98	11	>700
2	127	12	180
3	113	13	189
4	98	14	189
5	210	15	244
6	167	16	230
7	258	17	127
8	265	18	139
9	210	19	265
10	280		

It must be noted that Lysimeter 11 had not reached methanogenesis 534 d after the drums had been sealed. At the time of filling, 18.64 kg of the 80 kg mixture could not be fitted into the drum. Although the shredded refuse had been randomly taken from a pile of mixed refuse, the material for this drum appeared to have a large proportion of light, bulky waste such as plastic. The density of the waste in the lysimeter and possibly the biodegradable content, was thus much lower than in the other tests and this may have led to the retarded degradation rate. The composition of the waste was different to that in the other drums and was likely to have affected the rate of degradation. Conditions became acidic with pH values between 5.9 and 5.6 and did not become methanogenic.

From the remaining tests, it can be concluded that co-disposal of anaerobically digested sewage sludge reduced the duration of the pre-methanogenic phases. However, it is insufficient to recommend the practice on sanitary landfills simply because the rate of decomposition was improved. Effects of co-disposal on the leachate quality and volume are equally important, especially in a water deficient-environment.

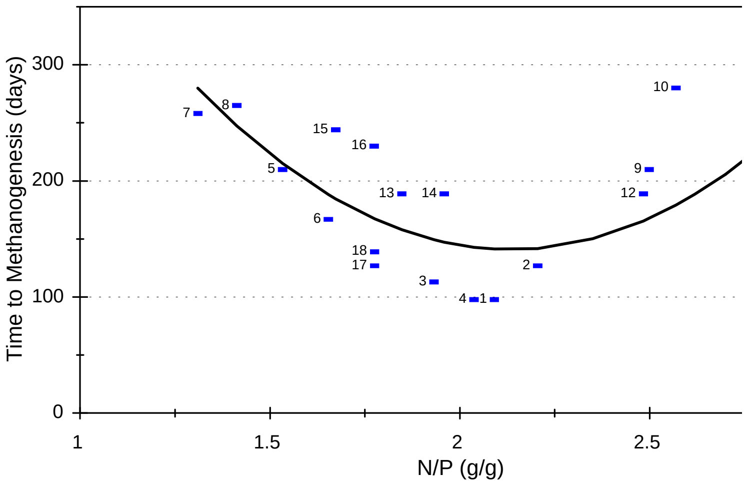
Although the sun-dried sludge was exposed to air after anaerobic digestion, the anaerobic bacteria were still present in the sludge. This may be inferred from the fact that the pre-methanogenic phase in the co-disposal lysimeters was shorter than in the refuse-only test. Methanogenic bacteria are spore-forming (Buivid et al., 1981) and during adverse conditions, such as exposure to air, the organisms become dormant. When conditions become anaerobic, the bacteria become active again. The differences in behaviour between the lysimeters containing sun-dried and belt-pressed sludge should thus not be attributed to changed microbial populations, but to altered levels of moisture and other sludge and refuse components.

Only a gradual decline in the duration of the lag phase could be observed as the initial moisture in the lysimeters increased (correlation to a straight line = 48%). By grouping the data according to the type of sludge used, this trend became more pronounced: for lysimeters containing belt-pressed sludge, a 14.2% increase in moisture halved the time taken for the mixture to reach methanogenesis. Only a slight difference was noted where sun-dried sludge had been used.

The duration of the acid-producing phase of the degradation process showed no response to altered levels of nitrogen or phosphorus when considering the individual effects of these nutrients on all lysimeters. Both the nutrients caused substantial changes in the lysimeters containing belt-pressed sludge, but no similar change occurred in the remaining lysimeters. Chemical supplements did not alter the rate of degradation compared to unaltered sludge (ANOVA, $\alpha = 0.05$), indicating that the nutrients in the sludge were used as readily as those from the chemical supplements.

A parabolic function (Eq. (1), correlation = 60.6%) relating the time taken for each of the lysimeters to reach methanogenic conditions to the nitrogen-to-phosphorus ratio indicates that an optimal C:N:P ratio exists. The ratio of 2.14:1 (N:P) appears to result in the greatest reduction in the acid-producing phase (Fig. 2), despite the fact that neither nitrogen nor phosphorus showed any effect when considered individually. From this study it appears that the optimal ratio is only a third of that proposed by De Baere and Verstraete (1984), which was 6.5:1 (N:P).

Figure 2
The relation between time to methanogenesis and the ratio of initial nitrogen and phosphorus of the waste mixtures



$$TTM = 1072.672 - 872.337 * (N/P) + 204.165 * (N/P)^2 \quad (1)$$

where:

- TTM = time to methanogenesis
- N/P = ratio of the initial mass of nitrogen and phosphorus in the lysimeter

By removing Lysimeter 10 from the estimation of coefficients, the correlation improved from 52% to 60%.

Previous research (Rees, 1980) indicates that increased moisture enhanced the rate of degradation, measured by the rate of gas production. However, the changed moisture levels were tested in isolation and possible interactive effects were not considered. Equation (2) relates the initial lysimeter contents to the duration of the lag phase (correlation = 84%) (Since the equation is four-dimensional, it is difficult to visualise or present in graphical form).

$$TTM = 801.124 - 490.136N_{\%} - 412.166\left(\frac{M_w}{M_s}\right) - 9.326P_{\%} + 233.951N_{\%}*P_{\%} \quad (2)$$

where:

- $N_{\%}$ = mass of nitrogen as a % of mass of solids
- $P_{\%}$ = mass of phosphorus as a % of mass of solids
- M_w = mass of water
- M_s = mass of solids

Interactions between parameters are important as indicated by the presence of the term which combines the nutrients, and moisture and nutrient effects are not simply additive. Although the coefficients for Eq. (2) are only based on 18 observations, the *form* of the equation substantiates the hypothesis that it is important to also consider the effects of interactions between parameters when considering the rate of degradation. This is supported by work of Buivid et al. (1981), where moisture alone did not increase methane yield, but only in combination with nutrients and buffer did moisture enhance methane yields.

Leachate quality

Concentrations of certain constituents are linked to the onset of methane production, for example, a sharp decline in the COD and some metal concentrations is evident as the leachate pH increases. The rate at which degradation occurs, especially prior to methanogenesis, affects the leachate collection and treatment measures. Co-disposal of sewage sludge and refuse alters the volume of water available for leachate formation as well as the levels of degradation products which can readily be transported from the landfill.

Digested sewage sludge contains the microbes responsible for acid formation as well as partly decomposed organic material in the form of simple organic acids. Because the methanogenic bacteria in the sludge are strict anaerobes, they become inactive during the early phases of landfill decomposition where they are exposed to air. They are slow to establish themselves and take 300 h to multiply whereas aerobic bacteria take only 20 min (Britz, 1993). It was expected that in the initial landfill processes co-disposal would thus enhance acid formation and make the conditions unsuitable for the methanogenic populations. Hojem (1990) predicted that leachate produced from co-disposal would have concentrations four times higher than from the refuse-only tests, even though the duration would be reduced. In the analysis, loads rather than concentrations have been used to account for variations in time and leachate volumes.

COD is an indicator of organic material in the leachate which is composed predominantly of simple organic acids. Sewage sludge addition reduced the cumulative removal of COD by leachate compared to the refuse-only lysimeter to about 40% of that released from the refuse-only drum, with the exception of Lysimeter 11 (not methanogenic 534 d after sealing the drums). The concentrations of COD in the leachate were also lower in the co-disposal tests compared to the refuse-only lysimeter, even prior to methanogenesis (Table 4). Lysimeters containing high loads of sewage sludge solids (Lysimeters 1 to 8) had cumulative COD removal values of only 30% of that of the refuse-only test. As the mass of sludge solids was reduced (Lysimeters 9 to 16), this effect weakened and the COD values showed a slight increase. This indicates that, although the refuse-only lysimeter became methanogenic at a similar time to some of the co-disposal tests - Lysimeters 7, 8 and 10 (see Table 3) - the leachate from co-disposal tests is more benign, even during the initial degradation processes and with only a small load of sewage sludge. This is supported by the findings of Stamm and Walsh (1988).

A range of COD concentrations from 6 000 to 60 000 mg/ℓ has been suggested for acid phase leachate (Ehrig and Scheelhase, 1993). In Lysimeter 5 a maximum concentration of 170 000 mg/ℓ was recorded, almost three times the expected value. This can partly be attributed to the shredded waste where a greater surface area is exposed to degradation than in unshredded waste. Ham and Bookter (1982) also found higher COD concentrations

TABLE 4
CUMULATIVE MASS AND CONCENTRATIONS OF LEACHATE CONSTITUENTS (534 d AFTER SEALING OF LYSIMETERS)

No.	Volume	COD	Ammonia (as N)		Phosphate (as P)		Nickel	
	(ℓ)	(g)	(g)	(mg/ℓ)	(g)	(mg/ℓ)	(mg)	(mg/ℓ)
1	29.121	233.19	55.58	1826.67	0.378	15.86	8.32	0.408
2	28.909	387.20	57.71	1911.14	0.357	15.12	14.15	1.063
3	30.726	335.75	46.82	1472.79	0.362	14.29	12.62	0.602
4	25.832	278.21	48.86	1777.83	0.251	12.76	10.62	0.687
5	9.902	293.12	36.12	3654.55	0.307	29.71	17.87	2.666
6	12.599	203.42	40.70	3299.09	0.200	18.91	21.68	2.785
7	8.031	470.23	19.76	2657.50	0.202	21.58	37.40	3.899
8	7.130	470.99	23.94	3331.11	0.124	19.19	36.04	7.468
9	16.943	595.37	30.35	1759.09	0.172	10.56	14.64	1.082
10	20.495	934.63	26.40	1271.09	0.331	18.45	21.35	1.572
11	12.524	1161.07	22.07	1753.20	0.141	11.54	21.19	1.923
12	20.341	463.77	23.51	1117.56	0.258	15.02	12.31	0.794
13	19.166	558.98	36.93	1904.05	0.297	18.08	40.54	2.900
14	16.293	438.05	32.20	2010.26	0.215	14.58	31.94	2.620
15	14.529	604.91	29.26	1990.00	0.243	18.91	47.63	4.371
16	21.366	695.42	25.35	1183.41	0.325	16.49	45.72	2.441
17	19.844	285.23	37.83	1924.21	0.270	15.38	20.72	1.361
18	17.803	219.45	34.58	1952.35	0.213	14.73	19.25	1.632
19	19.915	1216.65	19.74	1001.28	0.156	8.77	19.98	1.114

in leachate from shredded waste compared to unshredded refuse.

Because the lysimeters were used to simulate water deficient conditions, small leachate volumes were produced and this led to high concentrations of pollutants in the liquid. In many experiments large quantities of moisture passing through the waste bulk cause the leached constituents to be diluted.

Ammonia is a product of protein degradation and is also the form in which nitrogen is assimilated by microbes for growth. Under anaerobic conditions nitrogen exists as fully reduced forms of ammonia (NH_3) and ammonium ions (NH_4^+) (Baird, 1995) and nitrogen within the landfill will thus be converted to these compounds as oxygen levels decline. Ammonium levels have been shown to be independent of the changing landfill processes and various authors have noted a gradual increase followed by fairly constant concentrations of ammonium in leachate (Ehrig and Scheelhase, 1993).

Rees (1980) argues that nutrients are not limiting in the landfill since leachate contains compounds of these nutrients. The mass of nitrogen and phosphorus leached from the lysimeters containing sewage sludge was higher than from the municipal solid waste alone (Refer to Table 4). However, the shortened acid-producing phase indicates that the nutrients in the refuse were not at optimal levels for degradation.

In the lysimeter study ammonia nitrogen concentrations in leachate from the co-disposal drums was up to 3.6 times higher than from the refuse-only test. Higher nitrogen levels were initially measured in the sewage sludge than in the refuse and it is thus the source of the increased nitrogen in the leachate. Ammonium was leached from all lysimeters, indicating that nitrogen was present in quantities in excess of those which could be bound in the cell biomass during the growth of microbial populations in the landfill. However, the leaching of ammonia could limit the rate of degradation in the late stages of the landfill life. The presence of ammonia (or phosphorus) in the leachate is not necessarily an indication of an excess in availability and there are various possible explanations: preferential flow paths through the waste, the non-homogeneous nature of the waste and the varying rates of nutrient assimilation and waste degradation at different locations in the lysimeter or landfill.

No direct correlation could be found between the initial mass of nitrogen in the lysimeters and the ammonium nitrogen (concentration as well as cumulative removal) in the leachate. However, when the nitrogen, phosphorus and initial moisture were expressed as a proportion of the total mass of solids, Eq. (3) resulted in an 89.6% correlation between the actual and predicted ammonium concentrations in the leachate.

$$Amm[\,] = -2813.202 + 2179.614N_{\%} + 4257.448P_{\%} + 4697.493\left(\frac{M_w}{M_s}\right) - 8676.746P_{\%}{*}\left(\frac{M_w}{M_s}\right) \qquad (3)$$

where:

$N_{\%}$ = mass of nitrogen as a % of mass of solids
$P_{\%}$ = mass of phosphorus as a % of mass of solids
M_w = mass of water
M_s = mass of solids

Only a slightly lower correlation, 88.95%, was obtained from Eq. (4) which contains no terms for the initial amount of phosphorus. It can thus be concluded that phosphorus availability is not an important parameter in determining ammonium nitrogen concentration in leachate.

$$\mathrm{Amm}[\ ] = -3755.407 + 5290.219N_{\%} + 6619.154\left(\frac{M_w}{M_s}\right) - 6329.821N_{\%}*\left(\frac{M_w}{M_s}\right) \quad (4)$$

By expressing the nutrients and moisture in terms of the mass of solids, the terms in the equations become dimensionless and independent of the actual amounts of these parameters contained in the lysimeters. As the period of “constant” ammonia release progresses, the correlation coefficient should asymptotically approach its maximum value.

After 534 d of monitoring, co-disposal lysimeters had all released more phosphorus than the refuse-only lysimeter (See Table 4). Comparing the phosphate concentrations in the leachate, it was found that the type of sludge (i.e. sun-dried or belt-pressed) and the load-by-sludge type interaction affected the leachate quality (ANOVA, $\alpha = 0.05$). However, the chemical supplements to some of the lysimeters did not produce different phosphorus concentrations.

Leachate volumes also played a role in the mass of phosphorus leached: Lysimeters 1 to 4 had the highest cumulative removal of phosphorus and yet had the lowest overall phosphate concentrations and the highest leachate volumes. Phosphorus concentrations also depend on the rate of degradation so the average concentration is a deceptive parameter to use in these comparisons, but some of the lysimeters became methanogenic shortly after the first leachate was detected and the data could not be separated into “pre-methanogenic” and “methanogenic” leachate concentrations.

Nickel removal showed the effect of the sludge composition: the belt-pressed sludge (19% solids) caused a reduction in the mass of nickel leached, while the sun-dried sludge (84.5% solids) increased the nickel in the leachate, compared to the refuse-only lysimeter. Lysimeters 1 to 4 produced more leachate than the other lysimeters due to a high initial moisture content. It was expected that this would also cause more pollutants to be leached from the waste. Despite the high initial mass of nickel in the lysimeters with the high load of belt-pressed sludge (Lysimeters 1 to 4), the mass of nickel in the leachate was 40% lower than from the refuse-only lysimeter. Lysimeters which took longer to reach methanogenesis also had higher cumulative nickel removal.

Comparison of laboratory data and field observations

Field cells constructed at the Goudkoppies landfill in September 1993 were used to assess the effects of co-disposal subjected to daily temperature, precipitation and evaporation variability not modelled in the laboratory experiments. A refuse-only and two co-disposal cells, sludge-to-refuse ratios of 1:11.0 and 1:5.4, each containing approximately 400 t of waste were constructed using the same operational methods as the main Goudkoppies landfill (Röhrs, 1995).

Methane concentrations of 50% were recorded in the landfill gas from the two co-disposal cells after less than five months, with the refuse-only test reaching similar methane concentrations six weeks later (Röhrs et al., 1995). Since the site is in a water-deficient area, high methane concentrations were unexpected so soon after the cell construction.

After 15 months, no leachate had been obtained from any of the tests. This was despite above average rainfall for two successive summers and bunds around the top of the cells to minimise runoff (Röhrs et al., 1995). Modelling using HELP (Schroeder et al., 1994) indicated that the cell containing a high sludge load was likely to produce leachate after a year, however, after a further three months of summer rainfall no leachate was observed (Röhrs et al., 1995).

Based on these and the laboratory observations, it can be concluded that co-disposal accelerates the onset of methane production as compared to refuse-only tests. Although leachate was obtained from laboratory lysimeters, no leachate was obtained from any of the field tests which had been subjected to high temperatures and evaporation, and short, heavy downpours. Even the high load of sewage sludge containing only 20% solids (sludge-to-refuse 1:5.4) did not exceed the storage potential and evaporative losses at the site.

Conclusions

Co-disposal can provide a solution to some waste management problems. By enhancing the rate at which refuse degrades in the landfill, more waste can ultimately be deposited at the landfill site.

Anaerobically digested sewage sludge caused the refuse-sludge mixtures in the lysimeters to reach methanogenesis faster than the refuse-only lysimeter. Sun-dried and belt-pressed digested sludge altered the conditions in the lysimeters to favour methanogenic populations of microbes.

Increasing the mass of sewage sludge added to the refuse altered the moisture, nitrogen and phosphorus availability and increased the microbial populations. Lysimeters containing sun-dried sewage sludge had a lower initial moisture content than the other lysimeters.

The rate of degradation was shown to be dependent on the ratio of available nutrients, indicating that refuse is nutrient-deficient, even though nutrients can be detected in the leachate. Sewage sludge is complementary and provides nutrients to reach a more optimal balance in the landfill. By mixing anaerobically digested sewage sludge and refuse in ratios that will result in near-optimal nutrient proportions, the rate of degradation can be enhanced, making methane collection more feasible. The life of the landfill will be extended through more rapid decomposition of the waste.

Laboratory tests showed reduced cumulative COD removal and nickel concentrations in the leachate from the co-disposal lysimeters compared to the refuse-only test, but released leachate containing higher levels of both ammonium nitrogen and phosphorus.

Although more moisture was initially present in the lysimeters containing belt-pressed sludge and more leachate was generated from Lysimeters 1 to 4 than from other lysimeters, no leachate was detected in the field from co-disposal tests even though water had been ponded on the surface (Röhrs et al., 1995). The possible benefits to the full-scale landfill, choosing a sludge-to-refuse ratio of, say, 1:6 (such as Lysimeters 17 and 18) thus include: shortening of the acid-producing phase; no leachate formation; a hygienic, cost-effective means of disposing of sewage sludge without loss of usable landfill volume during placement; increased compaction density as well as earlier settlement and stabilisation than at a refuse-only landfill.

Acknowledgements

The work has been funded by the Foundation for Research and Development. The authors would like to thank the Scientific Services Laboratory of the Johannesburg Metropolitan Council for sponsoring and conducting all chemical analyses on the leachate and solids samples.

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