VOLUME 7

Deliverables to meet work plan objective 6: Collect data through instrumentation, field monitoring, and laboratory analysis that will enable the project team to assess the success of the project, the feasibility of this technology for other sites, and to enable the future design and operation of landfill bioreactors in Florida.

7.1 Work Plan Objective and Deliverables

Objective 6 of the project work plan was:

> *Collect data through instrumentation, field monitoring, and laboratory analysis that will enable the project team to assess the success of the project, the feasibility of this technology for other sites, and to enable the future design and operation of landfill bioreactors in Florida.*

The work plan identified the following methodology to meet objective 6:

> *The data collected through the instrumentation (Table 4.3; Table 4.9) and field/laboratory monitoring (Table 4.4; Table 4.10; Table 4.15) (see Volume 1 Bioreactor work plan for tables) for each site considered in this plan will be used to assess the success of the project. The rapid stabilization of the landfilled waste will be determined by measuring gas production, landfill settlement, and waste decomposition. Different treatment strategies will be evaluated for relative success in this manner.*

The deliverables identified in the work plan included:

- *Presentation of data in periodic reports.*

7.2 Assessment of Bioreactor Landfill Performance

In this volume, bioreactor landfill performance was evaluated through various field and lab tests. The tests discussed in this volume involve biochemical methane potential (BMP) assay of excavated waste samples, moisture contents of solid waste samples, waste decomposition rate, leachate quality and landfill settlement. Waste samples excavated from NRRL in 2001 and 2007 were analyzed for evaluating bioreactor landfill performance. Moisture content and waste composition and volatile solids results are discussed in Section 7.3.1. BMP assay results were used for waste degradation (Section 7.3.2) and waste decomposition rate (7.4). In addition to the settlement of NRRL, settlements of aerobic and anaerobic landfill bioreactor studies were compared and discussed as an indicator of landfill performance in Section 7.5. Details of test results and analytical methods are referenced at the end of each subsection. In addition to the bioreactor landfill performance test, research on estimation of engineering parameter such as hydraulic conductivity and air permeability is discussed in Section 7.8.

7.3 Waste Decomposition

One of the ultimate goals for operating a bioreactor landfill is to stabilize the landfill rapidly and to reuse the landfill site for other purposes. Thus it is critical to access waste decomposition properly as an indicator of landfill performance. Many approaches have been proposed to estimate waste decomposition. Among these analytical methods, biochemical methane potential (BMP) assay is known as one of the most precise and widely accepted methods. The BMP assay was firstly introduced by Owen et al (1979). Through measuring methane potential converted from biodegradation portion (mostly cellulose) of solid sample, waste decomposition can be estimated. As introduced in Volume 6, waste samples collected from a landfill were separated and divided into biodegradable and non-biodegradable fractions. Biodegradable fractions include paper and food waste. Plastic, textile, glass, stone and metal were categorized as non-biodegradable or refractorily biodegradable fractions. Because waste samples screened on No 40 sieve (sieve size, 0.475 cm) were too small to identify them, they were separated and categorized as "retained" fraction. Another fraction (size smaller than 0.04 cm) was separated and named as the "fine" fraction because most of this fraction consists of soil. Among all separated wastes, only biodegradable fractions (mostly paper), "retained" factions and fine fractions were taken into consideration as "biodegradable" portion of entire waste. Because the BMP assay is a microorganism mediated analysis, analytical variance should be considered a function of microbial activity (condition) and lack of nutrient. To minimize these variances, microbial seeds were activated in anaerobic digester and the amounts of waste samples used for BMP assay were normalized by volatile solids. Thus methane yields were reported as the volume of methane per volatile solids mass of waste sample added. This value can be converted to the volume per mass by multiplying percentage of volatile solids.

7.3.1 Waste characteristics

Moisture content. Moisture content distribution of waste excavated in 2001 is shown in Figure 7.1. With minor differences, the medians of measured moisture contents are in the range of 20 to 25% for entire depth of investigation, whereas the moisture contents of the 2007 samples were far greater than the 2001 samples. Except for the 0-10 ft depth, the median of the moisture contents was greater than 40%, and the median was greater as the waste depth was deeper. Since moisture was added from 10-20 ft depth of the landfill, the top of a landfill (0-10 ft) would be poorly wetted. The moisture contents of the 2007 samples were widely distributed in comparison to the 2001 data. The widest range of water distribution was observed from 20-30 ft depth whereas the narrowest distribution was found from 40-50 ft, the deepest. Moisture distribution measured by in situ instrument was illustrated in Volumes 3 and 5 and a master's thesis included in Appendix D*

Appendix C Thesis and Dissertation

- Jonnalagadda, S. (2004). “Resistivity and time domain reflectrometry sensors for assessinging in-situ moisture content in a bioreactor landfill.” Master's Thesis, University of Florida, Gainesville, FL.

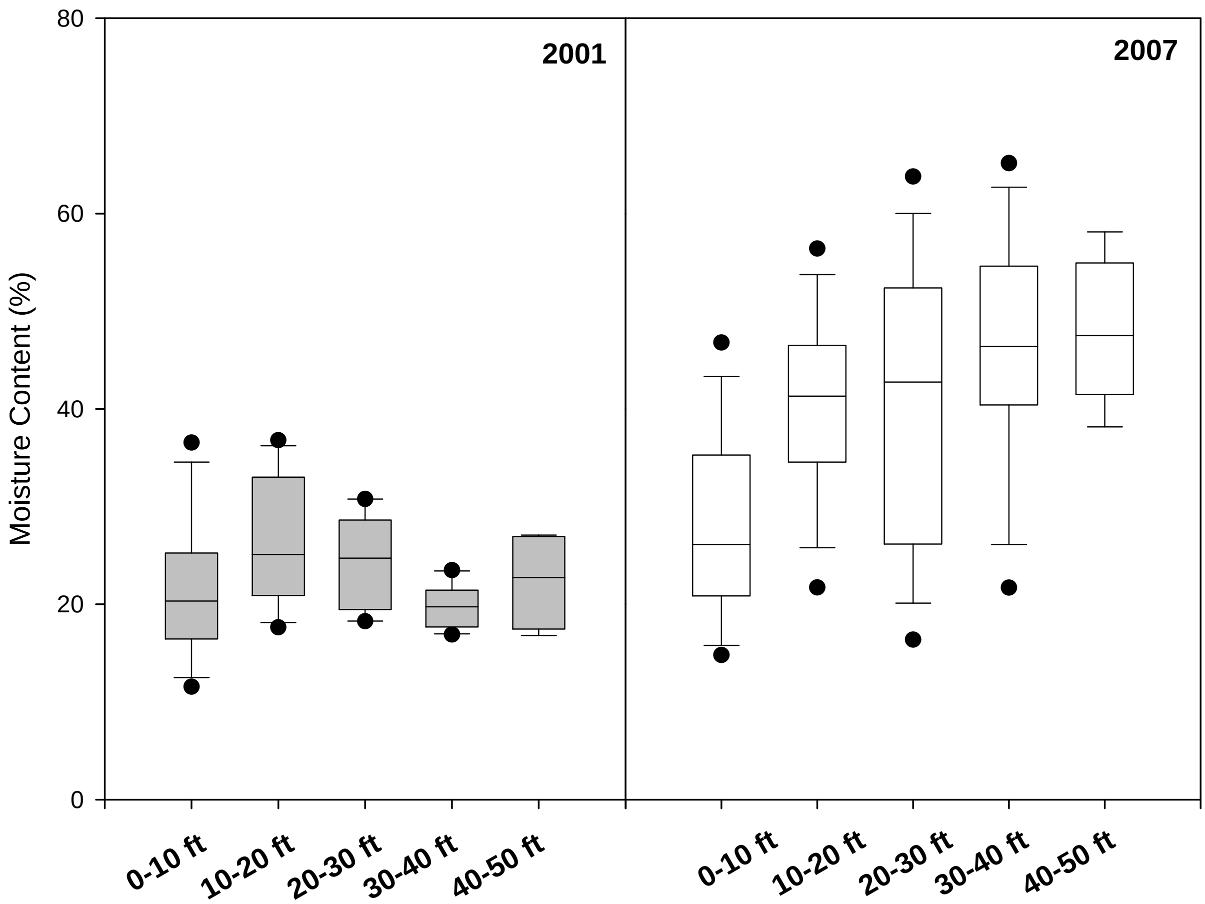


Figure 7.1. Moisture distribution of the waste samples excavated in 2001 and 2007.

Waste composition. Once waste samples were collected, they were stored in a -25°C deep freezer until they were processed. Before waste separation, all waste samples were dried in an oven for 2 days. Dried samples were then separated using a sieve shaker. Biodegradable and non-biodegradable materials were separated manually. As addressed in Volume 6, biodegradable waste (paper), “retained” fraction and fine fractions were further separated

Figures 7.2.1 and 7.2.2 show the average of waste composition between 2001 and 2007. A total of 50 (year 2001) and 57 (year 2007) samples were separated and values on the pie-charts

(Figure 7.2) represent the average values. The paper fractions, fine fraction and "retained" fraction of the 2001 samples were slightly reduced in comparison with 2007 samples. However, no significant differences could be found when comparing them using a single factor analysis of variance (ANOVA) method at 95% as confidence level.

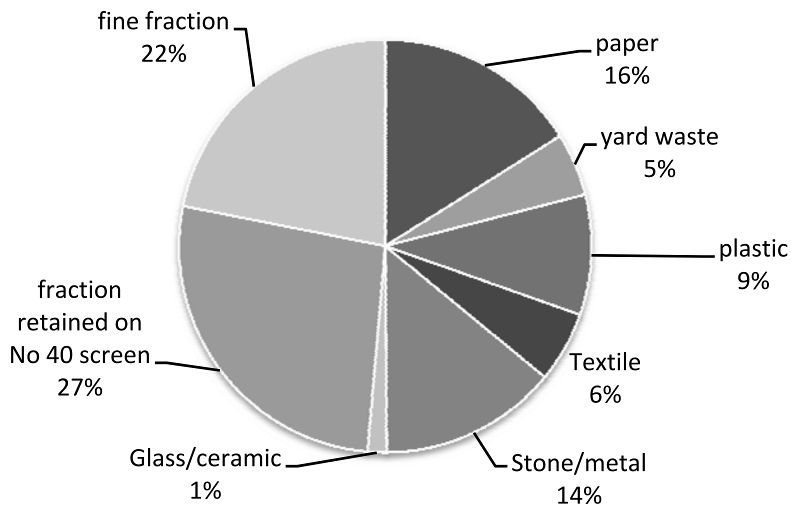


Figure 7.2.1. The composition of waste excavated in 2001

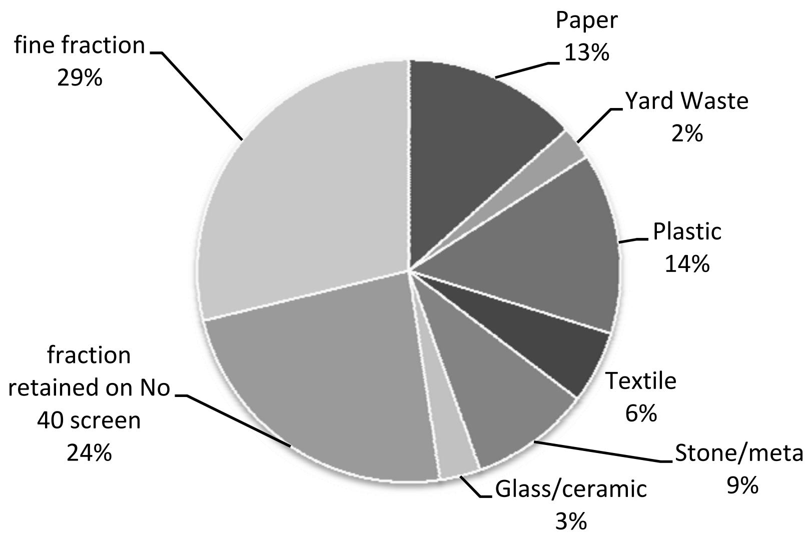


Figure 7.2.2. The composition of waste excavated in 2007

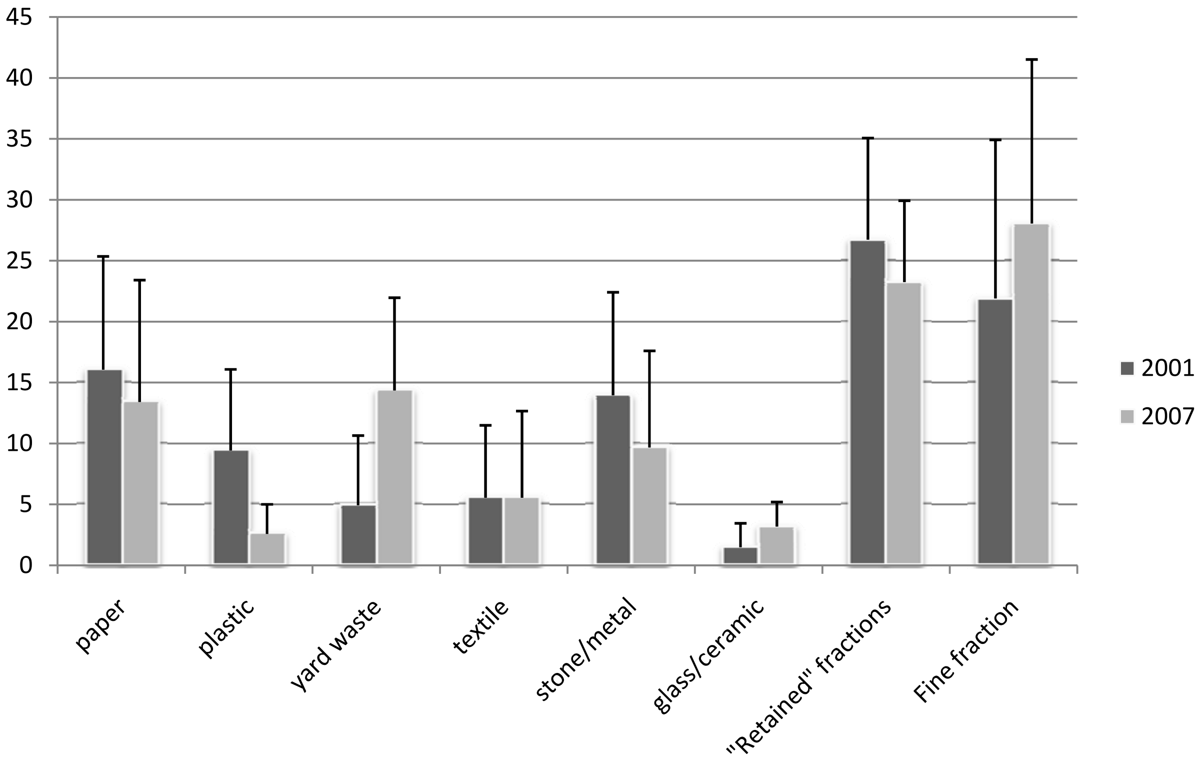


Figure 7.3. Comparison of waste compositions collected in 2001 and 2007

Volatile solids. The volatile solids (VS) fraction was measured by burning 50 to 100 g of dried samples at 550 ^{o}C for 2 hours. Volatile solids are the faction of solid sample considered organic matter and part of them were deemed as biodegradable fraction.

When comparing VS of 2001 and 2007 samples, no significant differences were observed in the fine and “retained” fractions; only paper fractions were reduced from 70% to 50% as shown in Figure 7.4. Although the volatile solids fraction can be used as one of the indicators of waste degradation, all volatile solids may not be biodegradable under landfill condition. Volatile solids can be divided into two categories; biodegradable and non-biodegradable. The biodegradable fraction is called as biodegradable volatile solids (BVS) and refractory volatile solids (RVS) for non- (or less) biodegradable fraction. RVS consisted of complex polymers such as lignin. Lignin can be decomposed or depolymerized in aerobic condition by fungi, but it is deemed to be non-biodegradable under anaerobic landfill conditions.

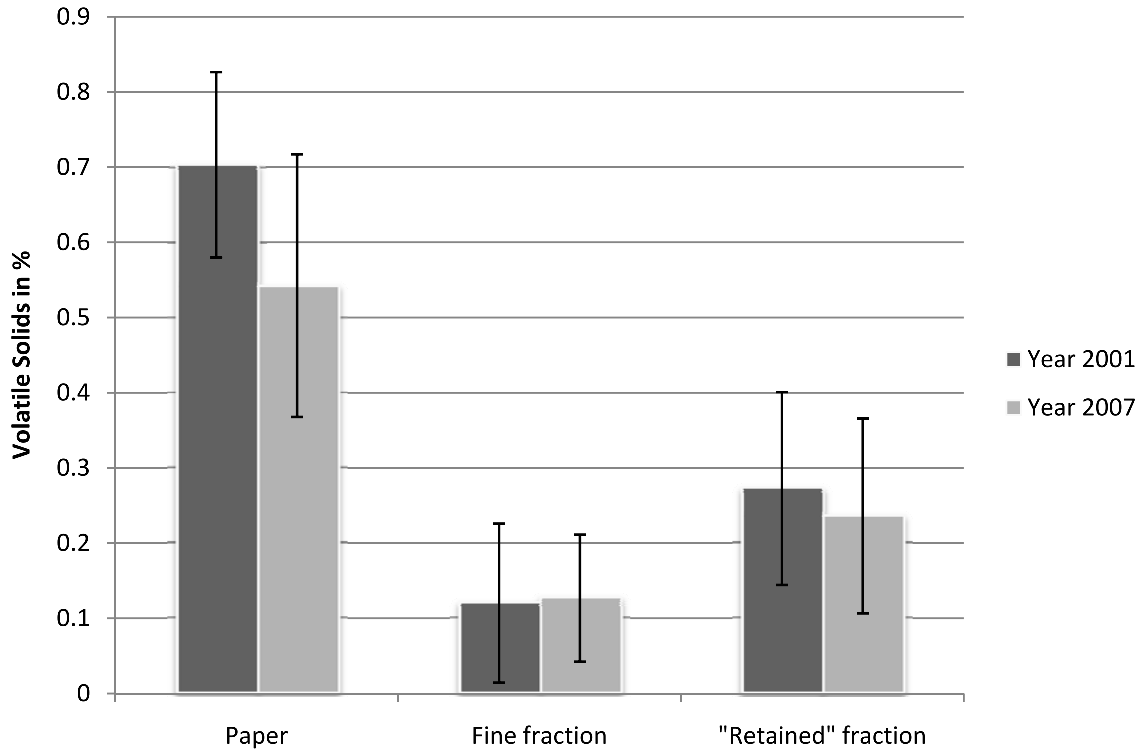


Figure 7.4. The comparison of volatile solids among paper, fine fraction and "retained" fraction between 2001 and 2007 samples

7.3.2 Biochemical methane potential (BMP) assay results

The biochemical methane potential (BMP) assay was designed to determine the biodegradable fraction of solid sample. There are several microbial decomposition steps needed to reach the final products in anaerobic condition, but a portion of carbon source used for maintaining microorganisms itself would be ignorable; it can be assumed that all biodegradable fractions may be converted into methane gas in methanogenic condition as follows:

$$C_6H_{10}O_5 + H_2O \rightarrow 3CH_3 + 3CO_2$$

Thus less methane can be generated as waste is degraded. To convert all biodegradable fractions into methane gas, 0.2g as volatile solids of waste are mixed with anaerobically digested seed and stored in 35^{o}C incubator. A period required for converting all biodegradable factions into methane would be 45 days to 60 days depending on biodegradability of waste. This method can be used for estimating toxicity and biodegradability of solid samples as well as BVS.

As addressed in Volume 6, solid samples excavated from NRRL in 2001 and 2007 separated and only paper fractions, fine fractions and "retained" fractions were taken into consideration as fractions containing biodegradable materials. Total number of solid sample collected for BMP assay were 50 and 60 in 2001 and 2007, respectively. Among 70 samples in 2007 samples, 48 samples were analyzed for BMP. After BMP assay completed, three fractions (paper, fine and "retained fractions) were combined and calculated total BMP in L_{CH4}/g VS $_{waste\ added}$ as follows:

$$\text{Total BMP } (L_{CH4}/\text{g VS}_{added}) = \frac{\mu_B \times VB \times WB + \mu_R \times VR \times WR + \mu_P \times VP \times WP}{VB \times WB + VR \times WR + VP \times WP}$$

Where, μ_X = methane potential of X fraction (B = paper, R = "retained" or P = fine fractions) in L/g VS, V_X = volatile solids of X fraction in %, and W_X = mass portion of X fraction (dimensionless).

Figure 7.5 shows the methane yield distribution between 2001 and 2007 samples versus number of samples in percentage. To compare methane yield distribution between 2001 and 2007 samples, methane yields were categorized into every 0.05 interval, and number of samples was normalized in percentage. Whereas methane yields of 2001 samples were concentrated around 0.15 to 0.3 L/g range, 2007 samples were inclined toward o to 0.1 L/g VS range. This indicates that 2007 samples were biologically degraded even though there were no significant differences in waste composition.

Differences of methane potentials would be more clearly observed by plotting on the box plot as shown in Figure 7.6. Methane yield results included in the "box" indicate the data fallen into 25th to 75th percentile. As shown in Figure 7.6, methane yields of 2001 and 2007 samples included within 25th and 75th percentile were completely separated. When comparing them using ANOVA test, there were significantly differences between 2001 and 2007 samples with 95% as confidence.

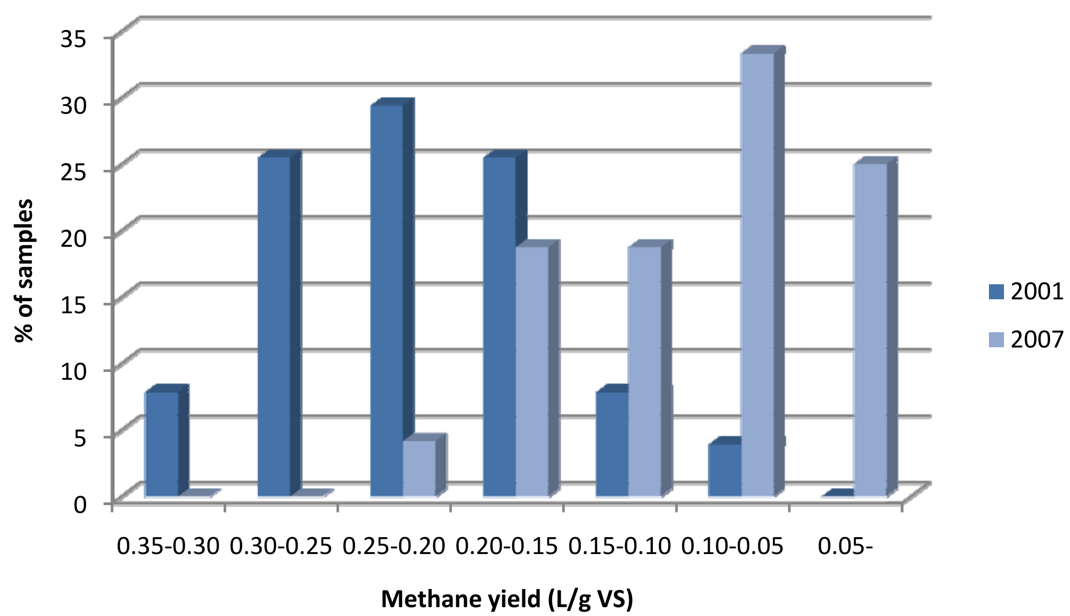


Figure 7.5. Methane yield distribution of 2001 and 2007 samples versus number of samples in percentage.

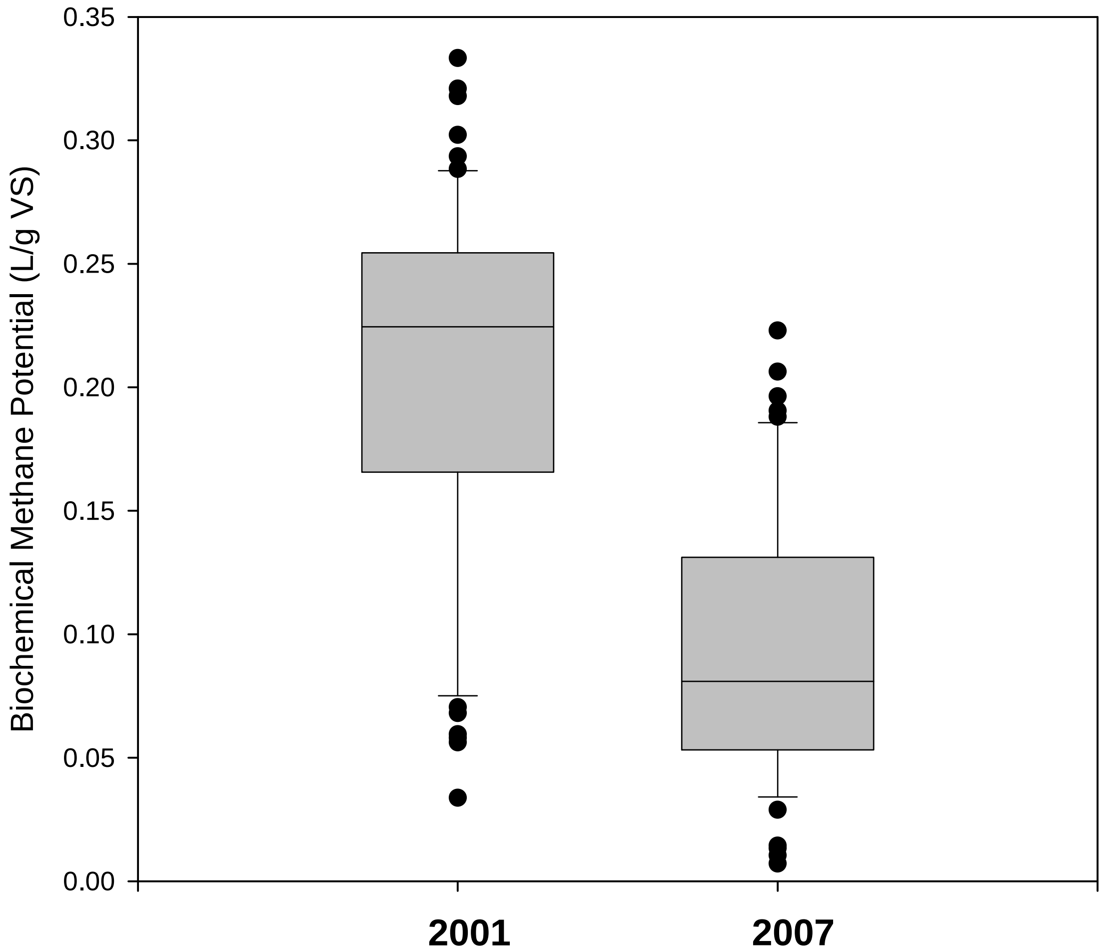


Figure 7.6. Biochemical methane potential assay results of waste excavated in 2001 and 2007

Changes in biochemical methane potential by depth. Biochemical methane potentials of waste samples with different depth were compared. For 2001 samples, average number of samples for every 10 ft depth was 10. For 2007 samples, waste samples were collected from 0-10 ft, 15-25 ft and 30-40 ft depth and average number of sample per each location was 16.

Figure 7.7 shows the methane yields distribution of 2001 samples by depth. Although there were some minor differences observed among samples, median of methane yields were not substantially different; they were in the range of 0.20 to 0.26 for all depths. When comparing methane yields of 0-10 ft samples (collected from top) and 40-50ft (collected from the most bottom) samples using ANOVA test, there were no significant differences with 95% confidence. It is implied that samples disposed of at the beginning of landfill construction were not well degraded while new waste was landfilled at the top of a landfill. According to the time line of landfill construction described in Volume 2, a gap between bottom and top would be 3 to 8 years.

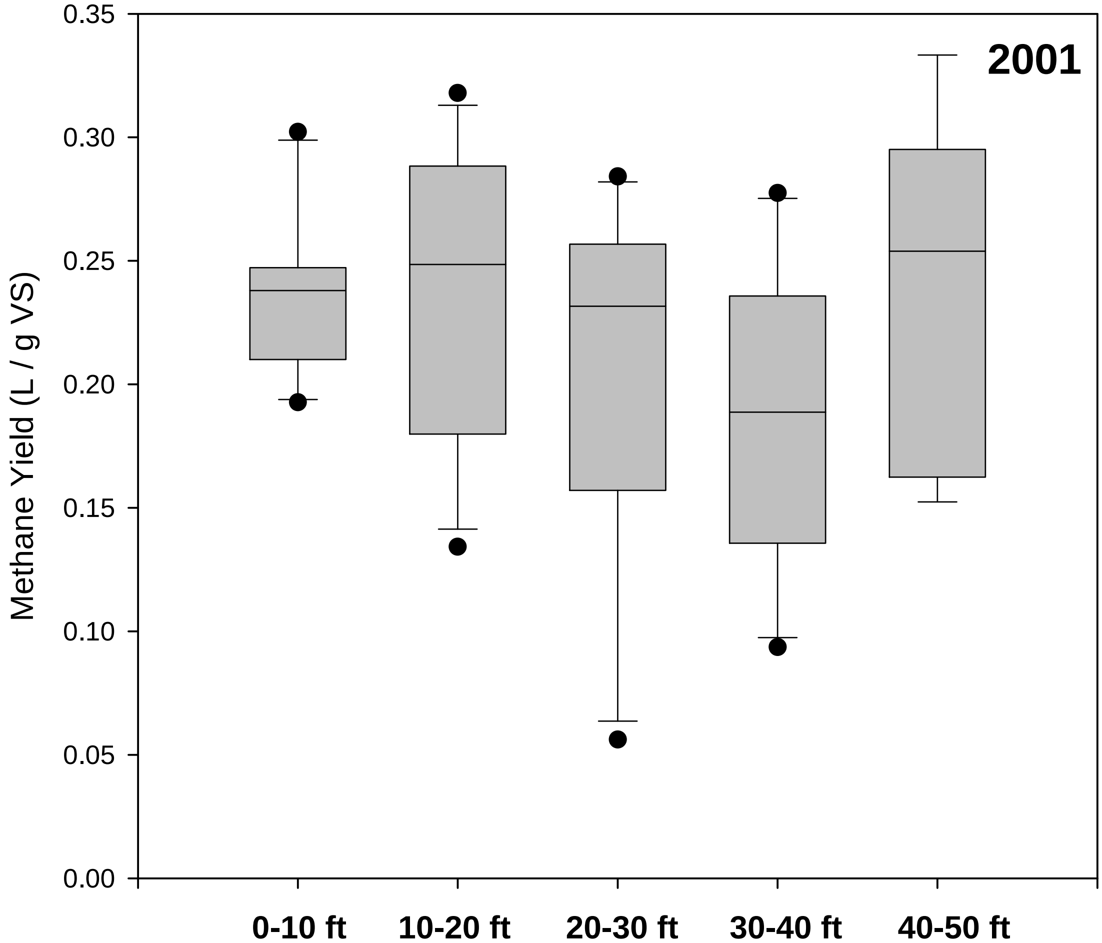


Figure 7.7 Methane yields of 2001 waste samples by depth

Methane yields of 2007 samples over sampling depth are depicted in Figure 7.7. Medians of methane yields were lowered as sampling depth increased, and the range of 25th and 75th percentile also decreased with sampling depth. Figure 7.9 shows the differences of methane yields by depth between 2001 and 2007 samples.

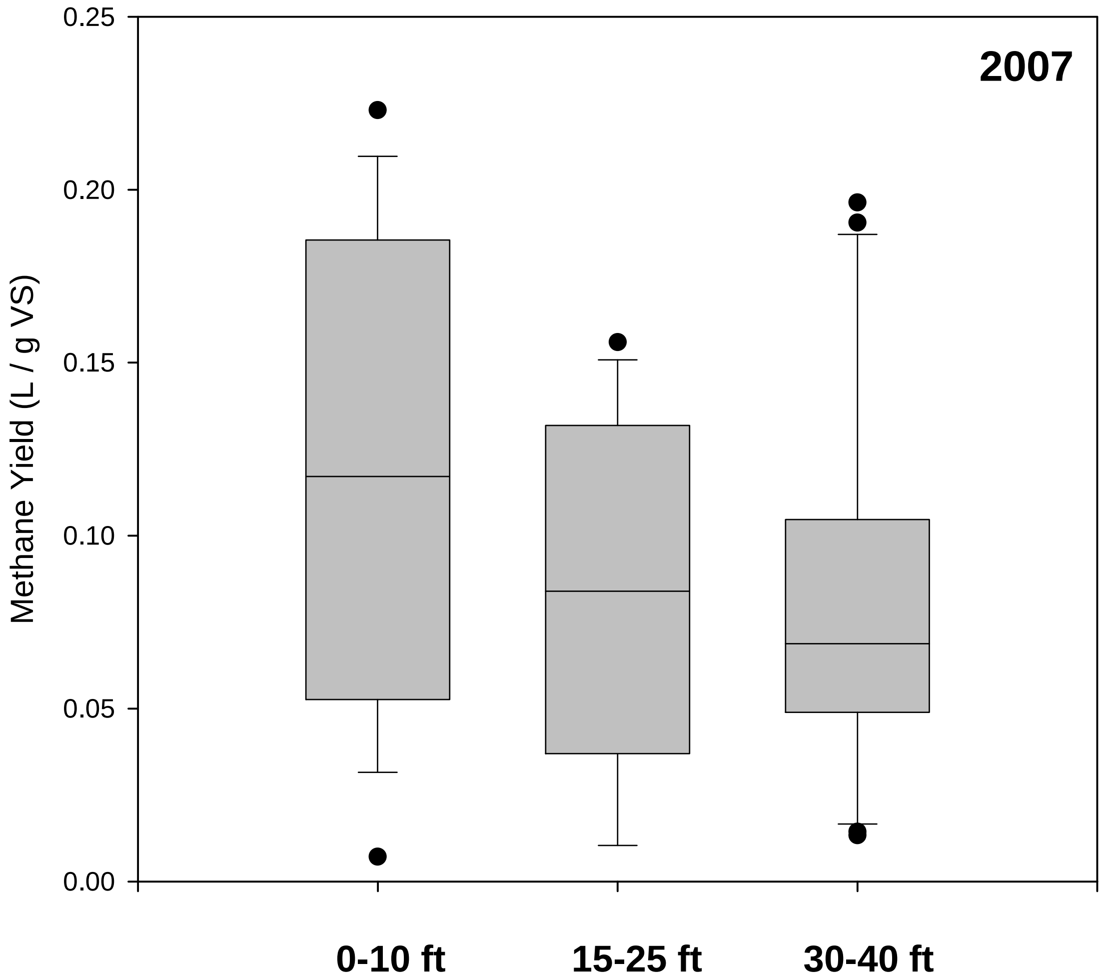
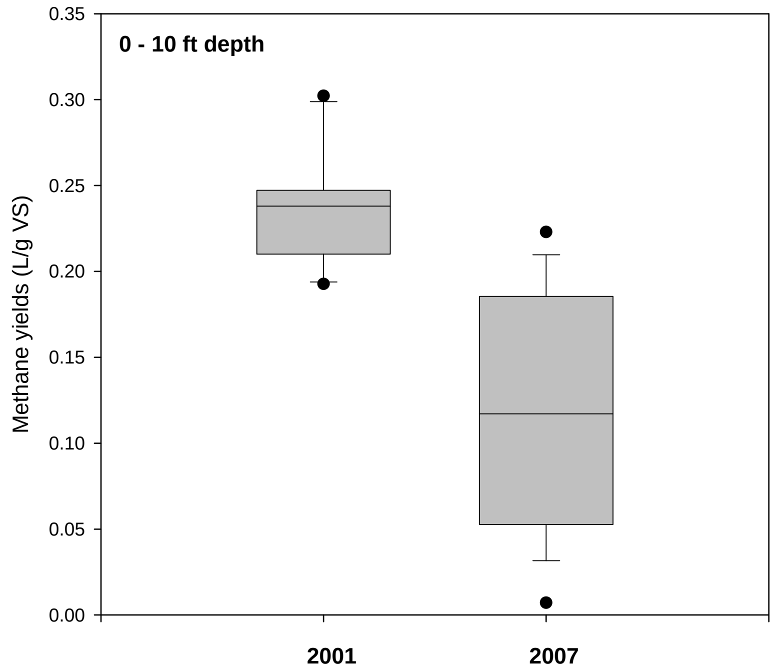
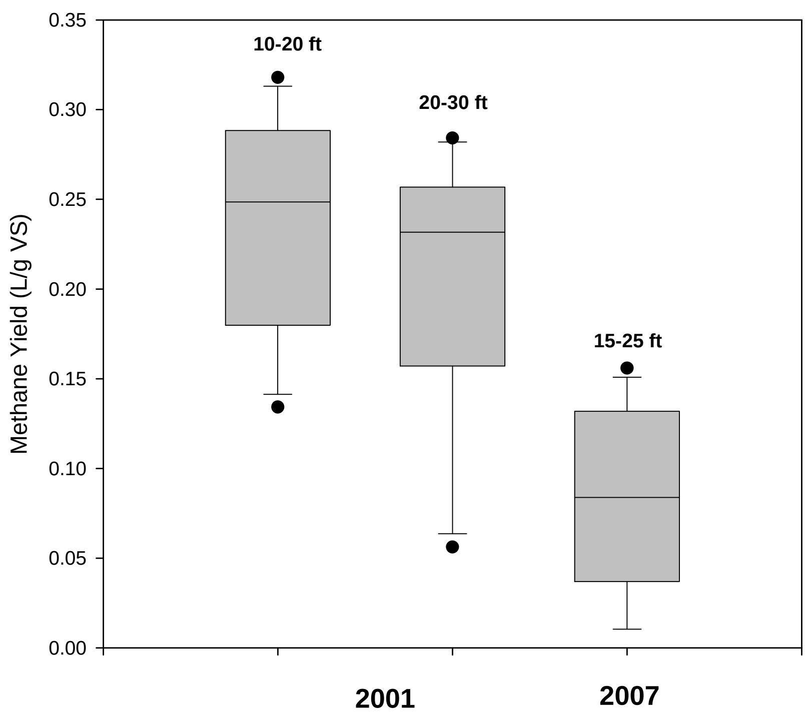


Figure 7.8. Methane yields of 2007 waste samples by depth

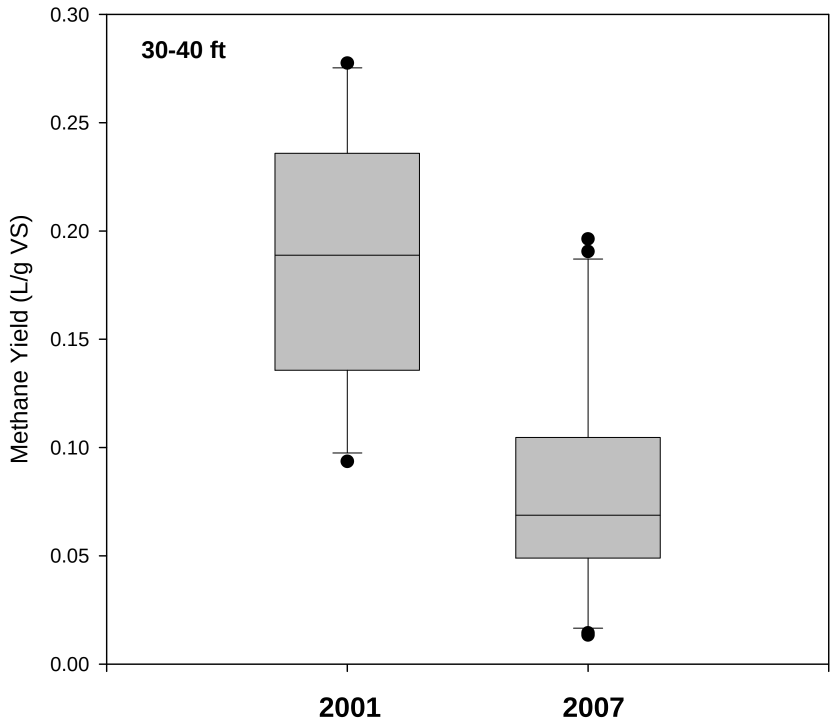


(a)



(b)

Figure 7.9 Comparison of methane yields between 2001 and 2007 waste samples with different depth; (a) 0-10 ft, (b) 10-30 ft and (c) 30-40 ft



(c)

Figure 7.9 (continued)

Effects of moisture contents on biodegradation. Figure 7.10 shows the relationship between moisture content and methane yields. Closed circle and open circle indicate 2001 and 2007 samples, respectively. Although ages of 2001 samples are approximately 3 to 8 years, methane yields of these samples were still great and moisture contents were relatively constant for entire depth. On the contrary methane yields of 2007 samples were substantially low and they were lowered with their moisture content as shown in Figure 7.10. This indicates that great portion of waste was decomposed as a result of leachate addition.

Figure 7.11 shows the relationship between methane yields and moisture contents of 2007 samples. Methane yields were lowered as moisture contents increased. It is noted that methane yields were not substantially changed when moisture contents reached 40%. Although water is necessarily present for waste degradation, excessively added moisture may not greatly affect waste decomposition. As addressed in the Bioreactor Operation Guide, it would be reasonable to set field capacity of waste as target moisture volume.

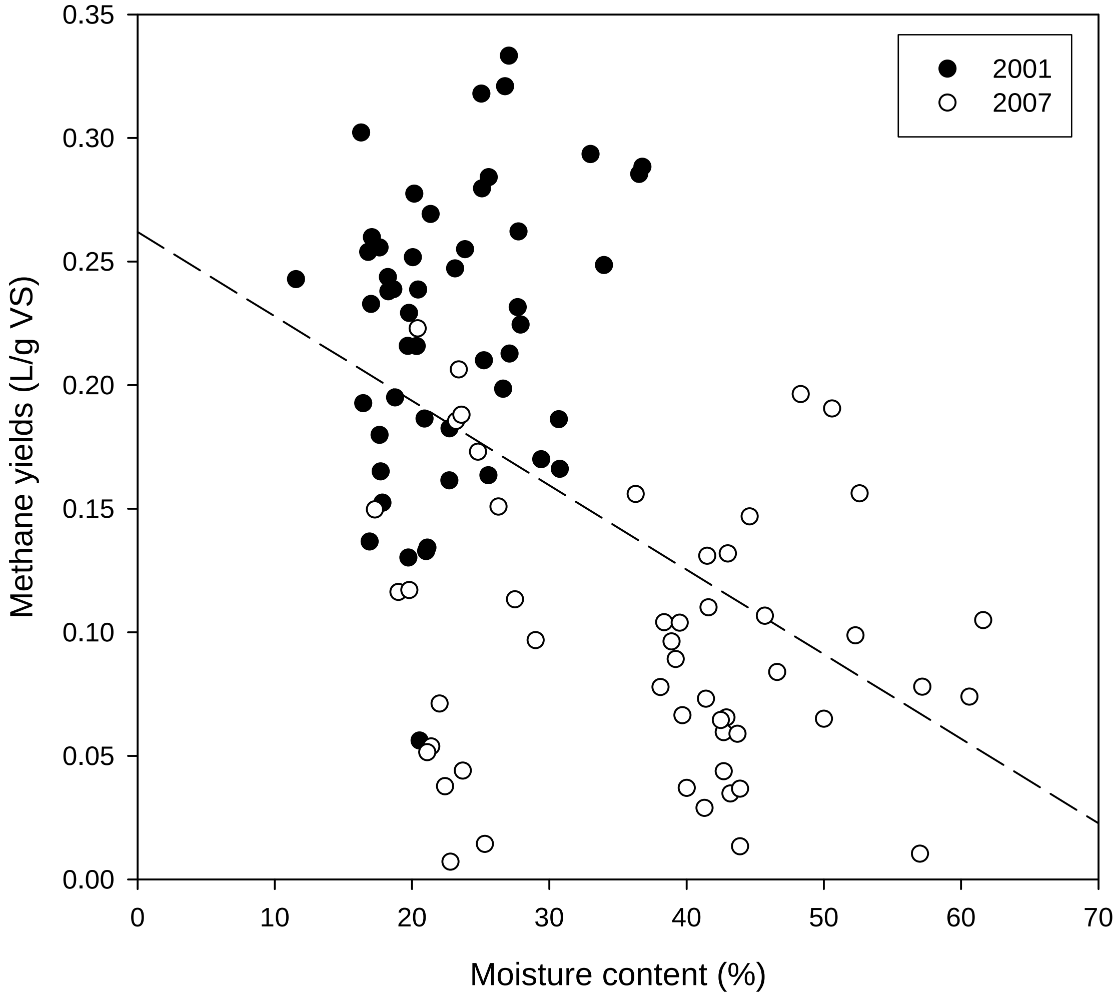


Figure 7.10. Relationship between methane yields and moisture contents.

7.3.3 Conclusions

In this research, the effects of applying leachate recirculation technique on waste decomposition were evaluated through moisture contents and BMP assay. Solid samples were excavated from NRRL landfill before and after moisture addition and characterized them for waste composition, moisture contents and methane potential. As a result of moisture addition for last 6 years, moisture contents of waste were substantially increased and it leaded to enhance waste decomposition.

Moisture contents of waste samples collected in 2001 did not vary by depth. Comparing methane yields between waste samples collected from top (0-10 ft) and bottom (40-50 ft), no significant differences were found from them. This would a typical 'dry-tomb' of traditional sanitary landfill. It would not be expected to see great enhancement of waste decomposition for a few years. On the contrary, methane yields of solid waste collected in 2007 were substantially low. These analytical results successfully proved the effects of leachate recirculation techniques on waste decomposition.

Another analytical results show that waste decomposition was strongly associated with moisture content; it was greatly enhanced at high moisture content. However, effect of moisture contents on waste decomposition was not substantially critical when moisture contents were greater than 40%. These results indicate that excessive moisture addition may not substantially enhance waste decomposition. This research suggest that moisture would be necessarily added to a landfill for the purpose of keeping waste wet condition, mass (e.g., nutrient and microorganisms) transfer and leachate treatment.

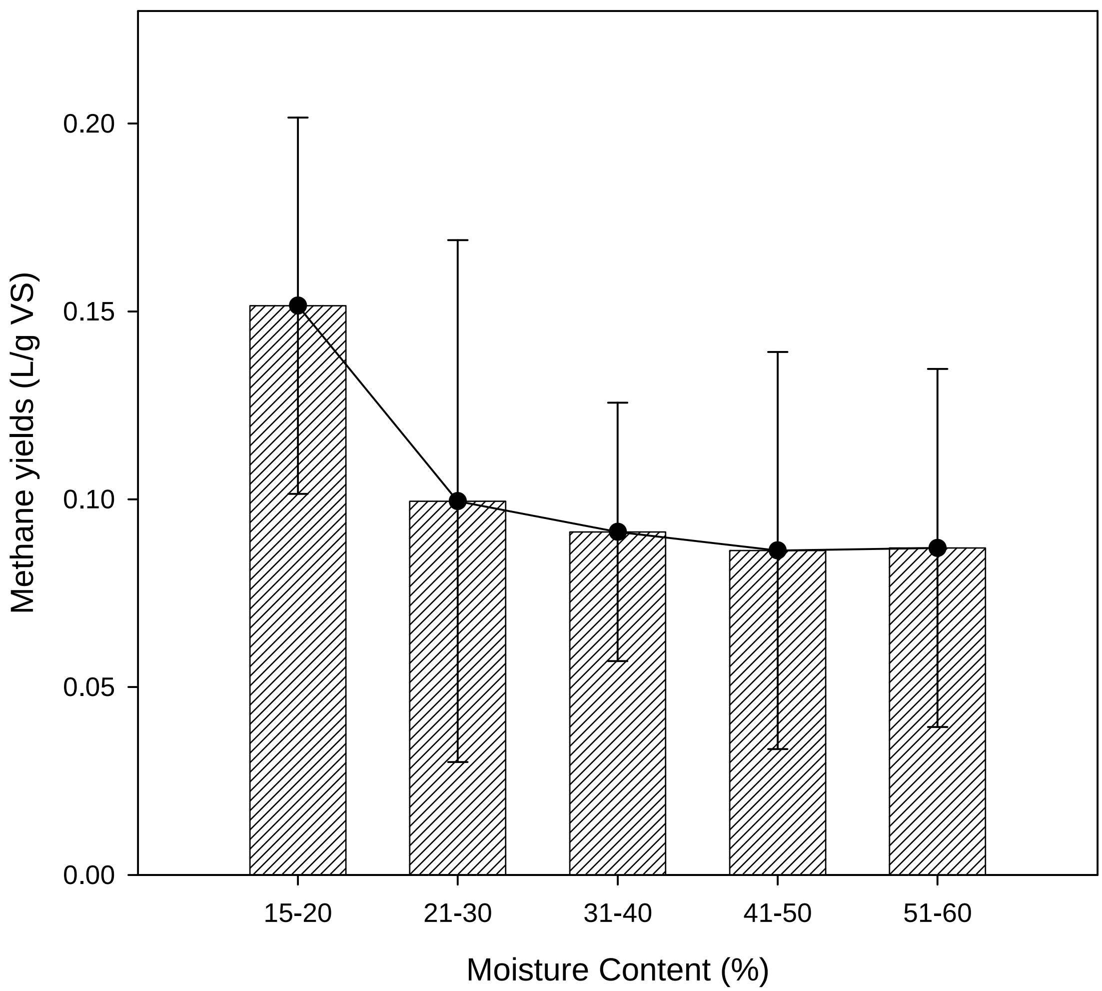


Figure 7.11. Relationship between methane yield and percentage of moisture contents of 2007 samples.

7.4 Waste Decomposition Rate

The first-order waste decomposition model has been widely adopted to predict landfill gas generation. Although some parameters used for the models need to be site-specifically defined, recommendable values for methane generation rate constant (denoted as k) and methane generation potential (denoted as L_o) were provided by USEPA. A value of 100 m^3/Mg waste, methane generation potential, is appropriated for most of landfill, but k value may vary depending on precipitation; a default of 0.04 $year^{-1}$ is used for area exceeding 64 centimeter (25 inches) of annual precipitation and 0.02 $year^{-1}$ is used for dry area. Depending on moisture content and technologies applied to a landfill, k value may increase even an order of magnitude higher than that of a conventional sanitary landfill. Fauor et al. (2006) estimated waste decomposition rates of three selected wet landfills using US EPA LandGEM model, and they were in the range of 0.11 to 0.21 $year^{-1}$. Default k value of LandGEM model for a bioreactor landfill is 0.7 $year^{-1}$ (USEPA, 2003).

In previous studies (Durmusoglu et al., 2005), gas production measurements have been used to indirectly estimate decomposition rates. However, due to the inevitable escape of gas (through the surface and sides of an active landfill as well as through the leachate collection system), gas capture even at its best is inefficient (Zison, 1990). Although gas can be collected from a landfill under protection liners, it would be difficult to collect gas from only leachate-treated area. Consequently, the estimation of gas production using gas generation is difficult.

Alternative method of estimating decomposition rates would be to use gas generation potential of solid waste samples excavated from an active landfill. Waste samples excavated from a landfill have been explored to evaluate landfill performance and to characterize landfill condition (ref). Among the analytical parameters of waste samples, methane potential has been heavily explored because it can provide the direct evidence for waste degradation under the given conditions (Owens and Chynoweth, 1998; Mehta et al., 2002; Ozkaya et al., 2004). With the methane potential (including L_0) and age of excavated waste, waste decomposition rate can be estimated. Despite its usefulness, there are only few studies available to access waste decomposition rate using excavated waste (Ozkaya et al., 2004). However, a few excavated samples they used may not be enough to estimate waste decomposition rate representing test area. Due to heterogeneity of waste, it is necessary to assess waste decomposition rate through numerous samples with different age.

Waste Decay and Gas Generation. A large fraction of municipal solid waste (MSW) is considered *biodegradable*. This includes components such as food waste, paper, and yard waste. The organic portion of such fractions is often estimated by measuring the material's volatile solids (VS) content—a term widely used in the characterization of biological waste treatment systems (Tchobanoglous, 1993). However, due to the presence of recalcitrant materials such as lignin, only a portion of the volatile solids content degrades under normal biological treatment conditions (Stinson and Ham, 1995). The biodegradable fraction of a material's volatile solids content (BF) may be defined as:

$$BF = \frac{M_{BVS}}{M_{VS}} = \frac{M_{BVS}}{M_{BVS} + M_{RVS}}$$

where M_{BVS} represents the mass of biodegradable volatile solids (the VS content biologically converted to gaseous end products) and M_{RVS} represents the mass of refractory volatile solids (that remains after treatment).

A common process calculation in landfill engineering involves estimating solid waste mass loss resulting from degradation and conversion to gas. Prediction of total gas production is relatively straightforward (based on previous research on biodegradability of waste in laboratory experiments and field measurements). For example, the USEPA default gas production potential for typical MSW is 100 m^3 CH_4 per Mg of total waste (USEPA, 1997). The rate of gas production, however, is more complex. One typical approach is to model the waste in a landfill as a series of discrete batches, each undergoing biological decomposition. The gas produced at a given time by the decomposition of each batch is then summed, and the total gas production from the landfill is predicted. The most commonly used model for simulating decomposition from a batch of waste is incidentally, the same first-order decay used by the USEPA in predicting gas generation at landfills nationwide. (EPA, 2005). Equation 2 represents the first-order relationship that models the conversion of waste mass into landfill gas.

$$M_{BVS,t} = M_{BVS,o}\, e^{-kt}$$

$M_{BVS,o}$ is the original mass of waste ultimately converted to gas, t is the time, and k is the first-order decay rate [t^{-1}].

G(t), the cumulative amount of gas produced from this decay, may be modeled as

$$G(t) = \alpha_{BVS}\ M_{BVS,o}\left(1 - e^{-kt}\right)$$

where α_{BVS} represents the volume (or mass) of gas produced from the decay of one unit of M_{BVS}. The above equation may be further manipulated to predict the instantaneous gas production rate G`(t):

$$G'(t) = \alpha_{BVS}\ M_{BVS,o}\ k\, e^{-kt}$$

Again, the USEPA currently prescribes a default gas generation rate coefficient, *k*, of 0.05 yr^{-1} (half-life of 13.9 years). This decay coefficient, present in equation 2, is the same term as the gas generation rate coefficient in equation 3.

The study reported here is based on BMP data for solid waste samples collected at a bioreactor landfill. Results of the BMP assays for samples of waste at different treatment times were used to estimate a decay rate.

Estimation of Decay Rates. To determine the decomposition rate of waste in the landfill areas sampled, ultimate methane yield from the BMP assays was selected to determine $M_{BVS,t}$. The resulting $M_{BVS,t}/M_{BVS,o}$ values were then fit using regression analysis. In order to perform the linear regression, equation 7 was first linearized via a log transformation. Thus,

$$y = -k(t - t_{lag})$$

where

$$y = \ln\left(\frac{M_{BVS,t}}{M_{BVS,o}}\right)$$

After regression analysis, a typical slope-intercept equation resulted.

$$y = mt + b$$

Back transforming yielded the following equation:

$$\left(\frac{M_{BVS,t}}{M_{BVS,o}}\right) = e^{(mt+b)} = e^{(m(t-t_{lag}))}$$

From this equation, the waste decay constant *k*, and the lag time t_{lag} were identified.

Waste generation of NRRL bioreactor landfill. To estimate waste decomposition rate, k, for NRRL, t_{lag} was determined when 2001 samples were collected. Because waste samples were collected at March 27 to April 8 in 2001, 4/3/2001, an average of these period, was determined as t_0. Waste samples collection in 2007 was conducted at August 13 to 29 in 2007. An average time gap between two sample events was 6.36 years. M_{BVS} was determined from BMP assay results as follows:

$$M_{BVS} = BF \times M_{VS}$$

Where,

$$BF = \frac{L_{CH4},\ \text{waste sample}}{g_{VS},\ \text{waste}} \times \frac{162\text{g, cellulose}}{3\times 22.4 L_{CH4},\ \text{cellulose}}$$

$$= [\text{Methane potential } (L_{CH4}/\text{g VS added}) \times \frac{1}{0.415}$$

$$M_{VS} = \text{waste sample mass} \times \text{VS, \%}$$

Where, 162 g is equivalent of molecular weight of 1 mole of cellulose. Since 1 mole of cellulose can be converted into 3 moles of methane, methane volume would be 3 × 22.4 L/1 mole of gas at 0^{o}C and 1 atm.

Since 2001 sampling day was set as an initial time, t_0, M_{BVS} of 2001 samples were determined as $M_{BVS,0}$ and M_{BVS} of 2007 samples were used for $M_{BVS,\,t}$. Waste decomposition rate, k, calculated from these values was 0.14 $year^{-1}$. Although the calculated k-value is lower than

EPA- LANDGEM model default, 0.7 year^{-1}, this value ranged in between 0.11 and 0.21 year^{-1}, a range of selected wet landfills (Fauor et al, 2006).

Waste decomposition rate of aerobic and anaerobic bioreactors. As introduced in Volume 4, aerobic and anaerobic bioreactor landfills were compared using 6-feet stainless steel lysimeters. The waste decomposition rates, k, of each lysimeter were estimated. The calculated waste decomposition rates are summarized in Table 7.1. For overall waste decomposition rate, initial mass of fabricated loaded and final dry mass of decomposed waste were used as M_0 and M_t. For one of the aerobic lysimeter (Aerobic - 1), waste already reached to the stationary phase at day 650, additional decomposition rate during exponential phase was added to Table 7.1. Waste decomposition rate obtained from the anaerobic lysimeters were similar to the k-value of NRRL bioreactor demonstration study (0.14 year^{-1}).

Table 7.1. Waste decomposition rates (k) of aerobic and anaerobic landfill simulation experiment

	Time(day)	Final mass (Mt) in gram	Overall k, (year $^{-1}$)
Aerobic-1	1098	6883.5	0.206 (0.326)*
Aerobic-2	377	8741	0.368
Anaerobic-1	1650	7462.8	0.119
Anaerobic-2	725	9258	0.162

* During log phase

Appendix C Thesis and Dissertation

- Faour, A. (2003). “First-order kinetic gas generation model parameters for wet landfills.” Masters Thesis, University of Central Florida, Orlando, FL.

Appendix D. Peer-Reviewed Journal Articles

- Townsend, T., Miller, L., and Kim, H., “Bioreactor Landfill Waste Decomposition Rate Determination Using Methane Yield Results for Excavated Waste Samples”, under preparation.
- Kim, H and Townsend, T., “Landfill Settlement Behavior with Waste Decomposition”, Under preparation

7.5 Leachate Quality as an Indicator of Bioreactor Performance

It would be hard to conclude that anaerobic bioreactor landfill technique may greatly improve leachate quality. Sometimes organic carbon and ammonia levels may increase as a result of leachate recirculation and enhancement of waste decomposition. Bioreactor landfill technique may reduce a period for acid phase, an initial stage of waste decomposition, and switch to methanogenic phase rapidly. Reduced periods of acid phase may result in minimizing heavy metal release that mainly happens at low pH. In this section, effects of bioreactor landfill technique on leachate quality changes are discussed.

Organic carbon. For NRRL leachate samples, organic carbon level was low at all period of investigation except two manholes. Once leachate injection started, an increase in organic carbon level was observed from manhole 7 and 8 as shown in Figure 7.12. The organic carbon levels were kept high for 1 to 2 years and were lowered again. Changes of organic carbon over time can be more clearly observed in changes in volatile fatty acid (VFA) (Figure 7.13). Volatile fatty acids are by-products of waste decomposition and they are converted to methane gas by methanogenic bacteria. Since they are directly used by methanogenic bacteria as a substrate, they can be used as an indicator to check methane bacterial condition; if methanogenic population is not enough to consume all VFA, VFA (especially acetic acid) level increases and results in pH drop. On the contrary, bacterial population is large enough to consume all VFA, no VFA can be detected in landfill leachate indicating that all landfill eco-system is well established.

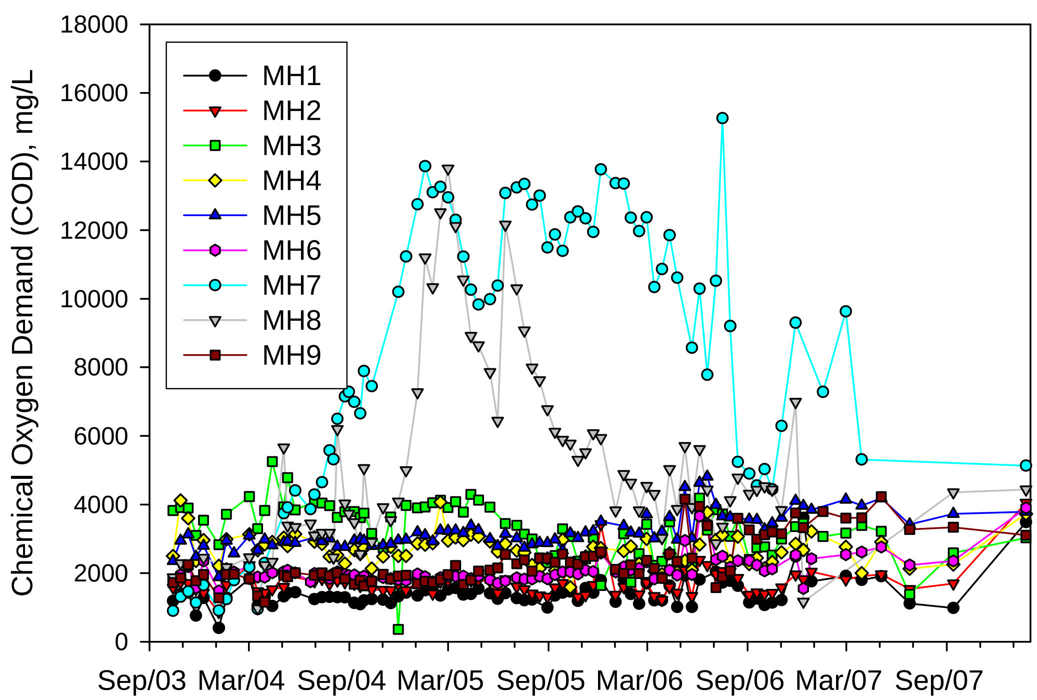


Figure 7.12. Changes in chemical oxygen demands (COD) over time.

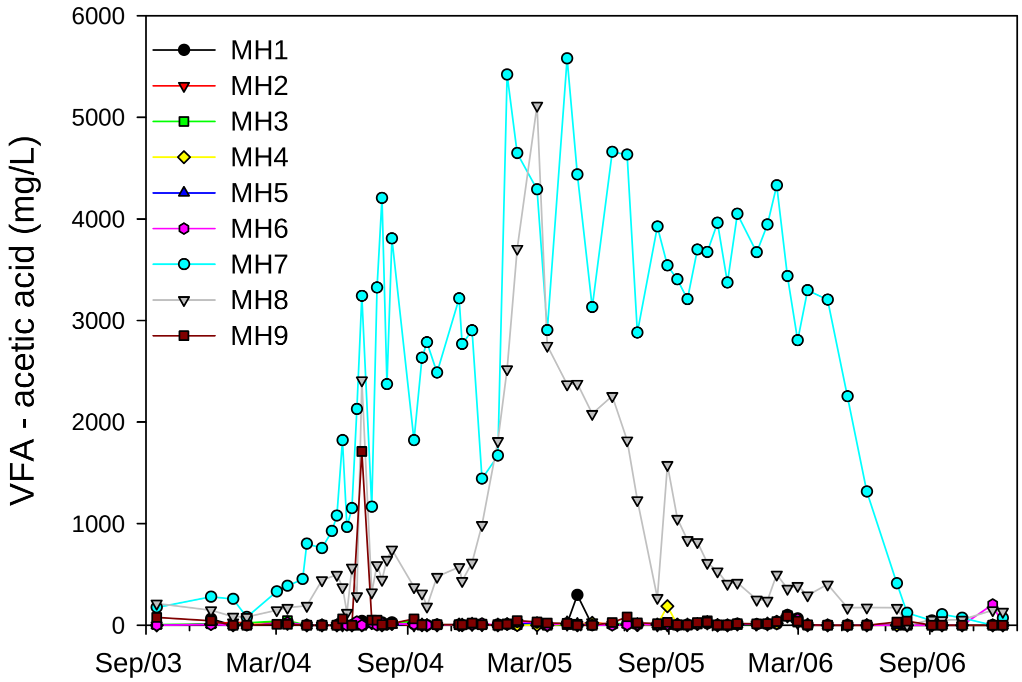


Figure 7.13. Changes in acetic acids of NRRL leachate over time.

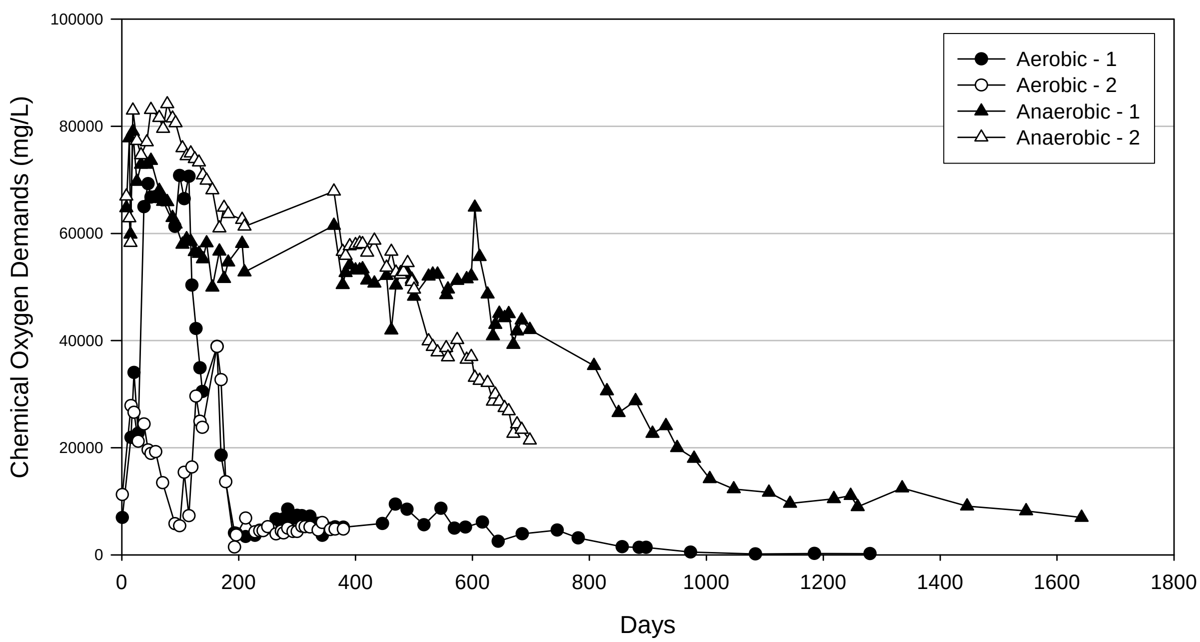


Figure 7.14. Changes in chemical oxygen demands (COD) of the aerobic and anaerobic lysimeters over time.

In Figure 7.13, No VFAs were detected from all manholes except for manhole 7 and 8. Acetic acid levels of manhole 7 and 8 were lowered again. Comparing between aerobic and anaerobic landfills, it is more clear that leachate quality can be improved in aerobic condition (Figure 7.14). Since

Ammonia. Because ammonia is the most reduced form of nitrogen, it would not be converted into other components in anaerobic condition. Main source of ammonia is landfill would be amino acid contained in food waste. For these reasons, ammonia level could be accumulative and increase as waste is decomposed. On the contrary, ammonia can be converted into nitrogen in the presence of oxygen through nitrification process. These differences are depicted in Figure 7.15.

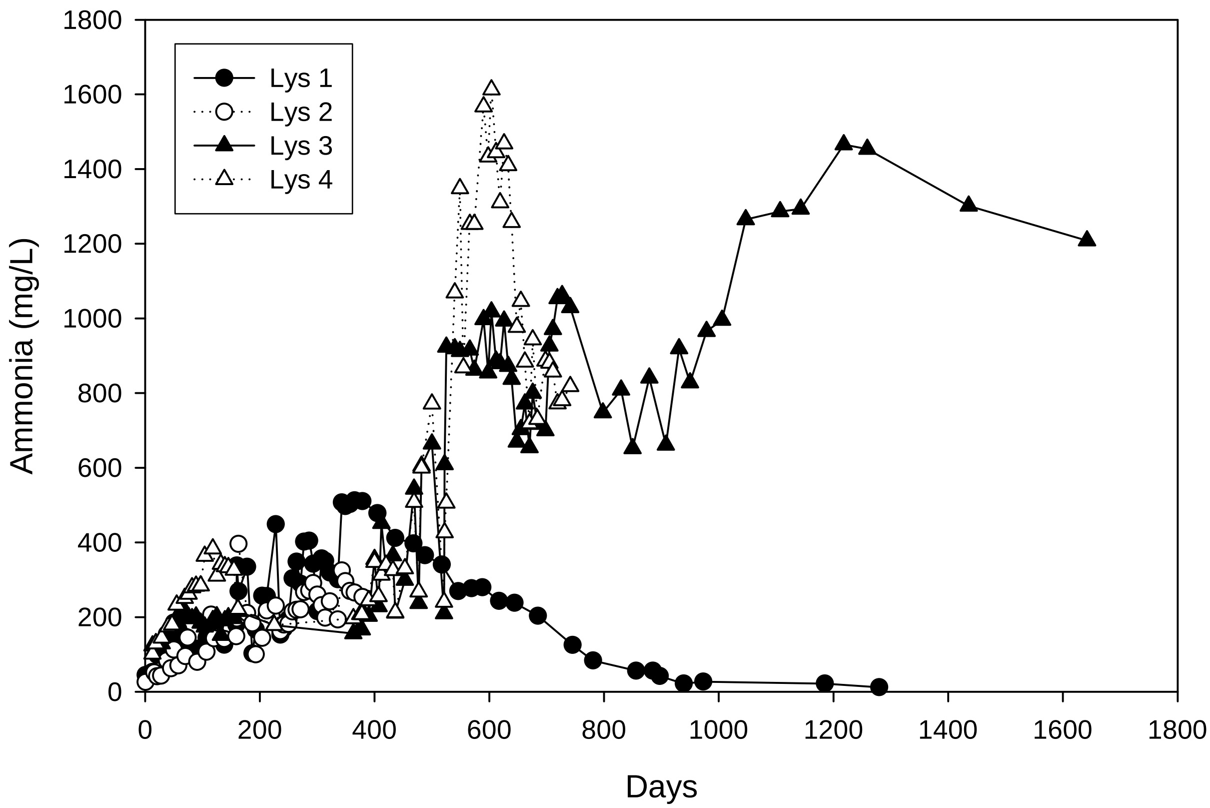


Figure 7.15. Changes in ammonia of the aerobic and anaerobic lysimeters over time.

Heavy metals. Heavy metal behavior may vary by pH and oxidation/reduction potential (ORP) of leachate. In anaerobic condition, most of heavy metals released out from their sources and increased heavy metal levels in landfill leachate. Once pH reached to neutral and ORP condition is lower (reduced condition), heavy metal concentrations in leachate are dramatically reduced. Without obvious heavy metal sources such as ash and pressurized wood products, commonly heavy metal would not be critical issue in anaerobic bioreactor landfill. However, unlike the other basic water chemistry parameters, heavy metal quality may not be always improved depending on their species. As addressed in Volume 4 and 6, high pH and (semi) oxidation condition drive some kinds of metals such as lead, copper, chromium and aluminum (Figure 7.16).

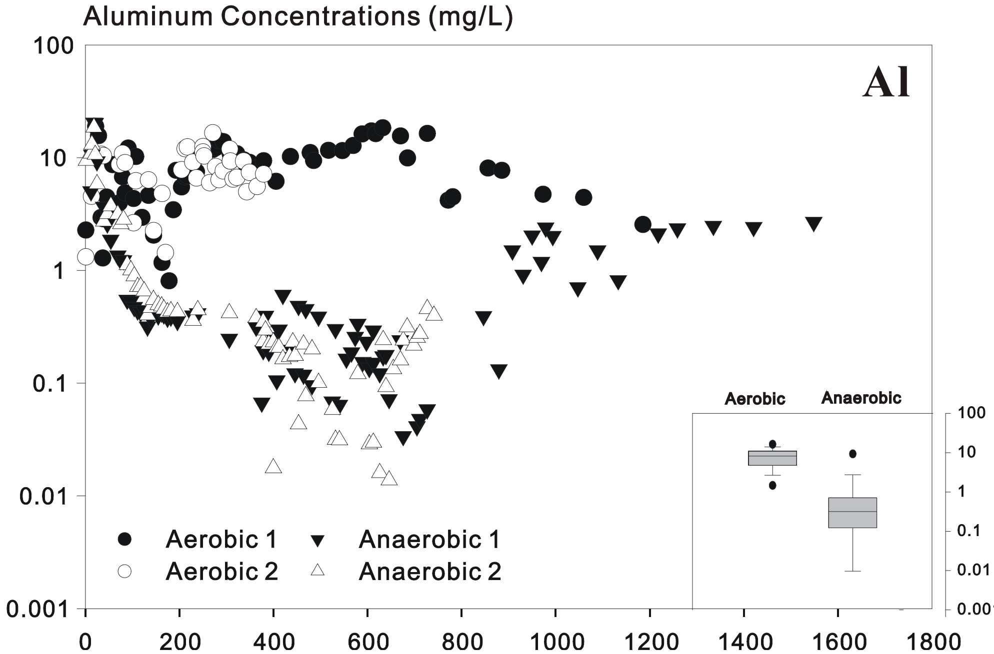


Figure 7.16. Changes in aluminum concentrations of the aerobic and anaerobic lysimeters over time

Differences between aerobic and anaerobic bioreactor landfills with respect of leachate quality. The largest differences between the aerobic and anaerobic landfills can be found from the enhancement of waste biodegradation. According to the lab scale lysimeter study, a period required for the aerobic lysimeters to decompose 90% of BOD was 160 days in the aerobic lysimeters while more than 700 days were required for one of the anaerobic lysimeters.

Tables 7.2 and 7.3 present the initial and final characteristics of the aerobic and anaerobic lysimeters. The data presented in Table 7.3 for the anaerobic lysimeters indicate that these systems were not stabilized yet. Water loss from the aerobic lysimeters was calculated using the water carrying capacity of the exit gas assuming that the gas was 100% saturated with water vapor. The overall performance of the aerobic lysimeters with respect to waste decomposition and leachate quality was substantially greater than those of the anaerobic lysimeters. However, the concentrations of sodium in the aerobic lysimeters were still too high to meet drinking water standards. The final concentration of ammonia in the aerobic lysimeters was also substantially higher than the criteria value of ambient water (0.897 at $30^{o}C$ and pH 8.0) (USEPA, 1999). Although large quantities of waste were decomposed, leachate of the anaerobic lysimeters still contained high concentrations of organics, ammonia and anions (Table 7.2). Leachate generated would be used for recirculation, but excessive volume of leachate must be treated at an on-site or off-site wastewater treatment plant.

Table 7.2. Comparison of initial and final characteristics of the aerobic lysimeters

	Initial		Final (1 year)	
	Lys 1	Lys 2	Lys 1	Lys 2
Water quantity (mL)	19,000	19,000	15,137*	15,582 (15,056*)
Dry waste quantity (g)	12,784	12,784	8,389*	8,740 (8,715*)
pH	5.7	5.7	8.5	8.5
COD (mg/L)	20,000	28,000	3,400	4,700
BOD	13,000	16,000	200	30
TOC	6,000	7,000	2,600	2,200
Ammonia	70	40	500	250
Fluoride	80	30	0	0
Chloride	200	130	1,200	1,700
Sodium	80	140	800	900

Table 7.3. Comparison of initial and final characteristics of the anaerobic lysimeters

	Initial		Final (2 years)	
	Lys 3	Lys 4	Lys 3	Lys 4
Water quantity (mL)	19,000	19,000	18,844*	18,833 (18,704*)
Dry waste quantity (g)	12,784	12,784	11,290	9,258 (8,997*)
pH	4.5	4.9	6.5	7.4
COD (mg/L)	65,000	67,000	42,000	24,000
BOD	48,000	62,000	14,000	6,500
TOC	26,000	27,000	12,000	5,600
Ammonia	120	100	1,000	800
Fluoride	1,500	1,400	460	200
Chloride	1,450	1,400	670	500
Sodium	2,000	2,000	4,800	3,800

Implications for full-scale application of aerobic landfill technique in terms of leachate quality. Unlike the lab-scale simulated landfill, it is extremely difficult to aerate an entire large-scale landfill. Highly compacted wastes make it difficult for an air stream to penetrate into the recesses of a landfill. Moreover, leachate characteristics resulted from air addition may be variable. As the analytical results have shown, leachate characteristics of lysimeters 1 and 2 were different, despite starting with the same waste stream and the same operational condition. The leachate characteristics of lysimeter 1 were similar to those of the anaerobic lysimeters during the first 180 days showing great concentration of organic matter despite air addition. This was because of the large anaerobic zones formed at the bottom of the lysimeter by improper air addition to the bottom.

Aerobic zones can be formed around air injection wells but anaerobic zones may still be present in the same landfill. However, coexistence of the aerobic and anaerobic zones can be used for recovery of acid-stuck 'sour' landfills. In this research, the air addition was conducted under the hypothesis that environments formed by aeration for a short period can be favorable to anaerobic microorganisms. A great amount of VFAs, which caused acidic conditions, may be rapidly consumed by aerobes living in a relatively wide pH range. Conversion of carbonic acid ($H_2CO_3^-$) to CO_2 caused by air stripping may increase the pH. With air addition with low flow rate, the anaerobic zones may be protected from oxygen intrusion because oxygen may be depleted by the respiration of aerobes. An additional technical strategy would be to add buffer such as lime along with air addition. Buffer added may increase the alkalinity concentration. Without high alkalinity, the pH of the landfill may decrease again when air addition is stopped. This could happen when methanogenic bacterial population was not enough to adapt to the new condition.

Appendix C Thesis and Dissertation

- Kim, H. (2005). "Comparative studies of aerobic and anaerobic landfills using simulated-landfill lysimeters." Ph.D. Dissertation. University of Florida, Gainesville, FL.

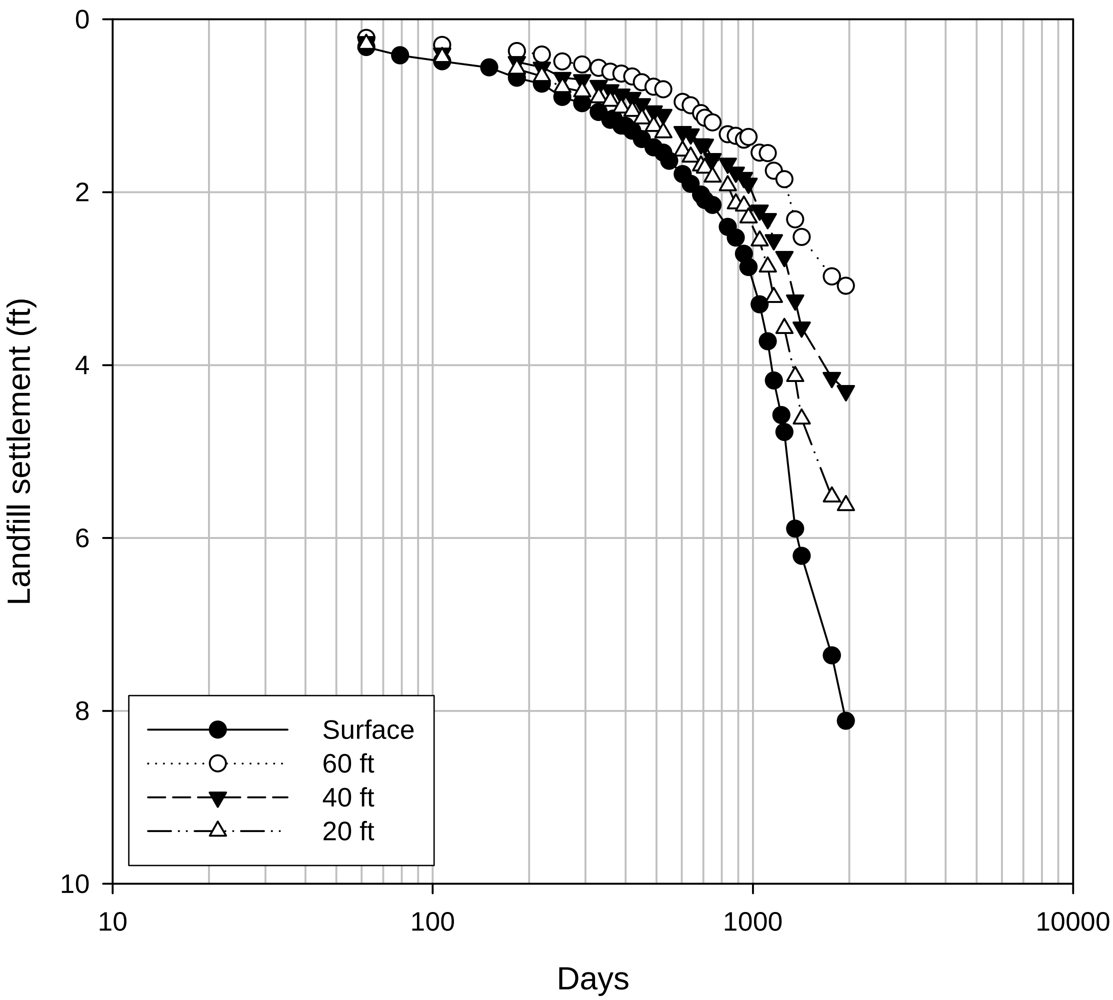
Appendix G Perioridic Technical Report

- 2006' New River Solid Waste Association (2006), "New River Regional Landfill Bioreactor Demonstration Project - Biennial report".

7.6 Settlement

The settlement of the landfill top surface and injection wells installed at different depths has been monitored since June 2002. Settlement measurements were performed periodically, approximately every six weeks, using a Magellan Ashtech Z-Surveyor with a GPS Field Mate. Ninety-four points were marked across the top of landfill to assess the settlement of the landfilled waste. Points chosen to survey the landfill surface include the concrete mogul fixed near each cluster of injection wells and monitoring wells for rigorous settlement analysis. For monitoring settlements of injection wells, the points marked on tees fixed to top of the leachate injection wells were also surveyed. Figure 7.17 presents the temporal settlements of landfill surface and injection wells. While Figure 7.18 shows the settlement behavior along with the leachate recirculation trend.

In addition to the routine settlement monitoring certain specific short term settlement data were also collected. The radius of influence of leachate recirculation around a well cluster was investigated over a period of time. Figure 7.19 demonstrates the radius of influence of leachate recirculation around a well cluster CM3. The three wells in this well cluster were injected with leachate simultaneously at the rate of 500 gallons per day each over a period of ten weeks starting July 22. 2003. The settlements were measured around this well cluster for over 69 weeks. The leachate injection in this well cluster shows the radius of influence of the leachate recirculation to be approximately 50 feet.



Day 1 = 6/24/2002

Figure 7.17. The settlement trend of the landfill surface, and three injection wells

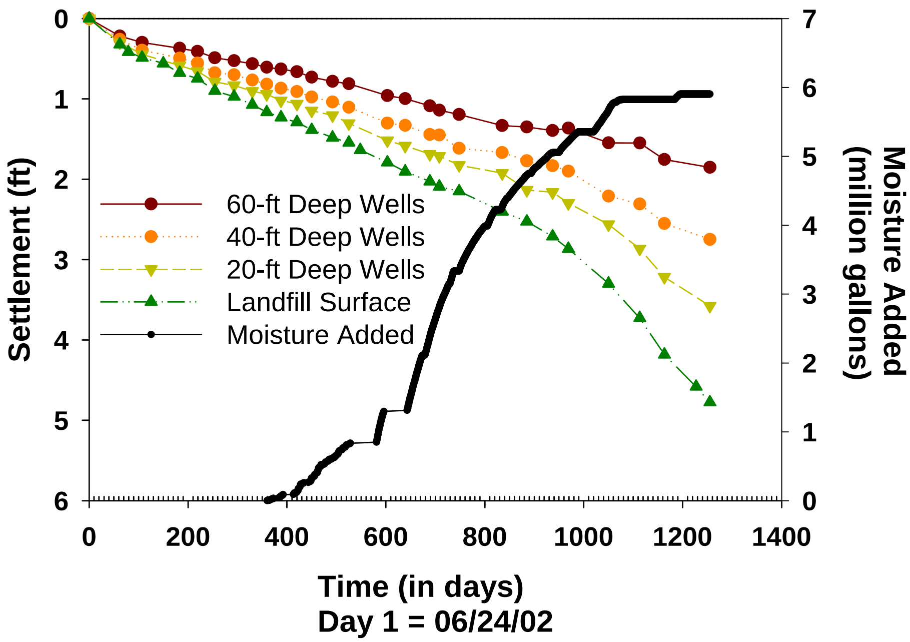


Figure 7.18. The settlement trend of the landfill surface, and three injection wells with respect to the leachate recirculation

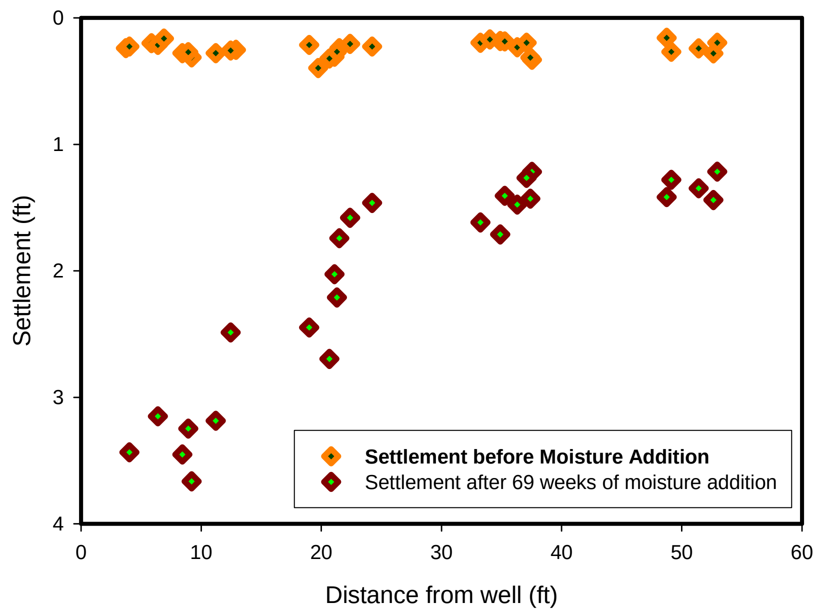


Figure 7.19. The settlement behavior around well cluster before moisture addition and after 69 weeks of moisture addition

Application of compression index and phase separate method to aerobic and anaerobic bioreactor landfills. Comparison study of aerobic and anaerobic bioreactor landfills with respect of settlement was conducted as discussed in Volume 4. Details about fabricated wastes and overburden pressure are described in Section 4.6.2 and 4.6.6. The measured settlement data were fit to several different relationships that had been previously proposed to model landfill settlement in the secondary phase (without consideration of condition with decomposition rate). These relationships included the modified secondary index proposed by Sowers (1973). Bjangard and Edgers (1990) developed the modified secondary index and explained landfill settlement mechanisms by separating the landfill settlement curve by different phases (phase separate method). Solving for parameter described on part of the relationship allowed comparison with other studies.

A modified secondary index (C_α') was used to describe settlement behavior of the aerobic and anaerobic lysimeters. Originally, the secondary compression index (C_α) was used to describe the secondary settlement for soil tests, but Sowers (1973) first applied this concept to landfill settlement. Since it is hard to estimate void volume in the field, the secondary compression index was modified. The modified compression index (C_α') is used to estimate the settlement that occurred after the first mechanical settlement. The C_α and C_α' can be expressed as follows:

$$C_\alpha = \frac{\Delta e}{\log(t_2 / t_1)} \tag{1}$$

$$C_\alpha{}' = \frac{\Delta H}{H_o \cdot \log(t_2 / t_1)} = \frac{C_\alpha}{1+e_0} \tag{2}$$

where Δe is the change in void volume; ΔH is the change in thickness of the waste layer; H_o is the original thickness of the waste layer; t_1 is the starting time of secondary settlement; and t_2 is the ending time of secondary settlement.

Bjarngard and Edgers (1990) proposed the phase separate method to describe the major causes of landfill settlement. They separated landfill settlement curve into two phases, $(C_\alpha)_{min}$ and $(C_\alpha)_{max}$ by slope. The first phase, $(C_\alpha)_{min}$, indicates that settlement occurs by mechanical interactions such as delayed compression of the refuse. Landfill settlement that occurs in the second phase, $(C_\alpha)_{max}$, is caused by both mechanical interactions and waste decomposition.

Figure 7.20 depicts the changes in settlement over time on a logarithm scale. The settlement trends were separated into two phases, $(C_\alpha)_{min}$ and $(C_\alpha)_{max}$ by the changes in slope. For the lysimeters 1 and 2, time differences (Δt) for $(C_\alpha)_{min}$ were only 50 and 30 days, while Δt for $(C_\alpha)_{min}$ were 210 and 410 days for the lysimeters 3 and 4. These results indicate that the settlement of the aerobic lysimeter mainly occurred by waste decomposition ($(C_\alpha)_{max}$ phase). The occurrence of biological activity during the settlement can be also confirmed by an increase in cumulative gas over time (Figure 7.20). For lysimeter 4, a rapid settlement curve was observed after a long lag period. It is noted that the depth difference (ΔH) of lysimeter 4 during the $(C_\alpha)_{max}$ phase is close to ΔH of lysimeter 1 during the same phase. This result indicates that the aerobic landfill may show the better settlement rate as opposed to the anaerobic landfill within a very short period of time. However, great settlement performance was shown in both aerobic and anaerobic lysimeters during the $(C_\alpha)_{max}$ phase rather than $(C_\alpha)_{min}$ phase. Therefore, it is concluded that waste decomposition plays an important role for landfill settlement under both aerobic and anaerobic conditions.

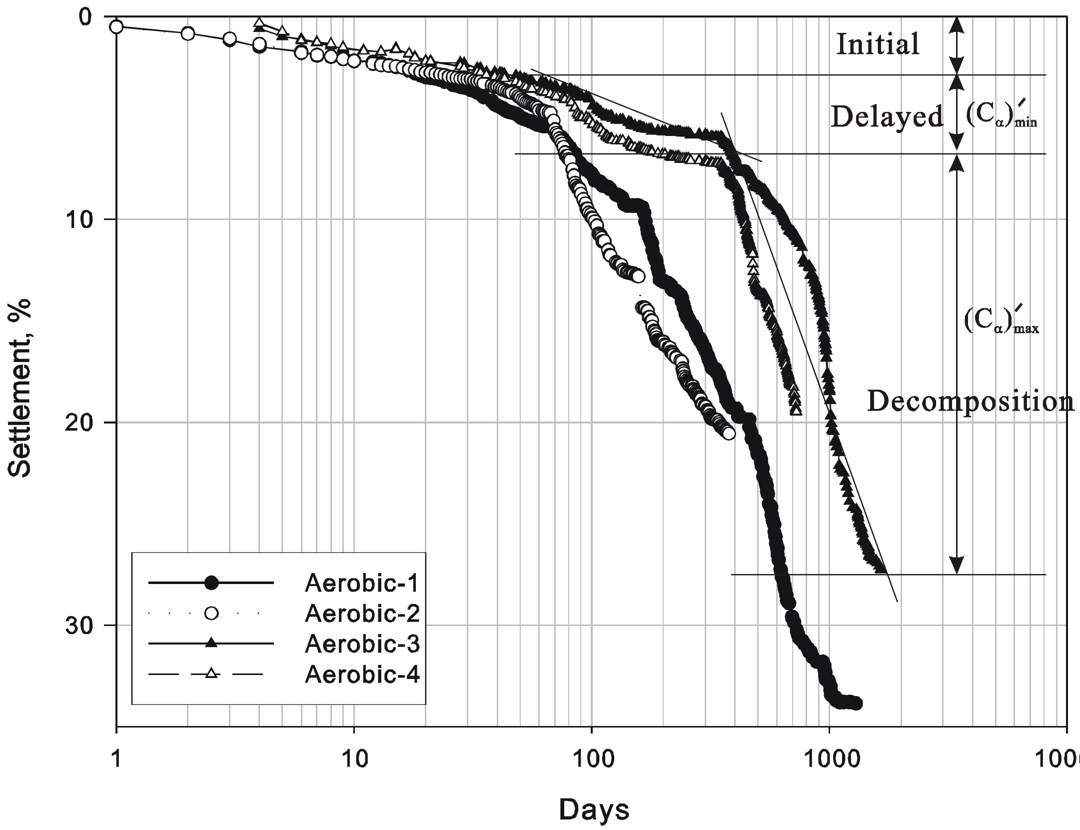


Figure 7.20. Settlement behaviors and compression coefficients of aerobic and anaerobic lysimeter over a period of time

Appendix D. Peer-Reviewed Journal Articles

- Kim, H and Townsend, T., "Landfill Settlement Behavior with Waste Decomposition", Under preparation

Appendix G Perioridic Technical Report

- 2006' New River Solid Waste Association (2006), "New River Regional Landfill Bioreactor Demonstration Project - Biennial report".

7.7 Feasibility of Air addition

7.7.1 Effect of continuous air injection on temperature

Results from air addition tests demonstrate a significant increase in temperature during air injection, and temperature in each case approached the permitted temperature limit. Details about this research can be found in Section 4.5.2 of Volume 4 and Powell's research included in Appendix D. Leachate recirculation at a location in previous months does not seem to slow the rate of temperature increase during air injection. Furthermore, the amount and duration of leachate recirculation used in the second phase of this test does not appear to be enough to avoid the temperature increase associated with air addition. The presence of moisture in the area surrounding an injection well is not necessarily an indication that the waste is saturated. It may be possible that if a greater volume of leachate is recirculated around an injection well, air could be injected for a longer period with a less dramatic temperature increase. In the case of this site, the landfill appears to be very insulated, and the heat generated by aerobic decomposition is not easily dissipated throughout the waste, causing localized hot spots to form.

Practical Implications. Short-term air injection tests were conducted to examine the zone of influence of injected air, using gas composition (CH_4 and CO_2) and pressure change as criteria. Results show that at an injection rate greater than 1.0 m3min-1, a significant impact (typically a 30% decrease in CH_4 and 15% decrease in CO_2) occurs at impacted monitoring points located within 17.4 m of the injection point. Injected air tended to flow in the north, south and west directions during testing, possibly attributable to the sandy clay cover material used during waste filling, the waste filling sequence, and potentially lower overburden pressure in the direction of the side slope.

Two separate injection tests were conducted to investigate the effect of different moisture addition scenarios on temperature increase related to air injection. Historical moisture addition appeared to have little effect on slowing temperature increases, and concurrent leachate addition had an impact on the moisture surrounding an injection point but was unable to prevent a rapid temperature rise. The ineffectiveness of historical leachate recirculation may be because an insufficient leachate volume was injected to the area or too much time between recirculation and air injection had elapsed. Simultaneous leachate and air addition may not have prevented temperature increase because not enough moisture was added to the area prior to air injection startup. The results obtained in this study demonstrate that the majority of air injected into a landfill travels radially.

We found that controlling temperature in a landfill during air injection is difficult; however, effective temperature control may be possible if a larger amount of moisture is added prior to air injection. Future studies should test other leachate addition scenarios, for instance adding a greater volume of leachate prior to air addition and adding leachate to multiple waste layers. The ability to control temperature during air injection is critical if this technology is to be implemented wide scale.

7.7.2 Landfill gas flammability

Landfill gas flammability would be another important issue to discuss the feasibility of aerobic bioreactor landfill. Because it would be not possible to make entire landfill aerobic, methane, oxygen and carbon dioxide, the three major gases, may be present inside of the landfill.

As discussed in Volume 4, the mixture of O2 and CH4 could be flammable when they are mixed with specific ratio in the presence of ignition source. This flammable gas mixture can be formed start and stop air injection into a landfill.

Landfill gas compositions (case study). The examples of the flammability potential of landfill gas (LFG) mixture formed by air addition can be found from elsewhere. Figure 7.21 shows the distributions of methane and oxygen concentrations over the flammability chart when air addition started or ended. The data points in Figure 4.6 were reconstituted on the basis of the published data (Read et al., 2001; Lee et al., 2002; Kim, 2005). In many cases, CH4 and O2 concentrations of landfill gas were distributed over the region III. These gas mixtures were formed within a relatively short period of time after air addition started or ended. According to the landfill gas data reported by Lee et al. (2002), relatively high concentrations of CH4 and O2 were shown during the extraction. These gas mixtures resulted in an increment in flammability potential. The authors explained that high concentrations of CH4 and O2 were dictated by air inflow from outside because of LFG extraction landfill at a greater rate than natural production. It is also noted that O2 concentrations in LFG could increase unexpectedly during operation according to the LFG data reported by Read et al. (2001). This indicated that gas monitoring needs to be uninterruptedly performed to prevent from gas explosion during whole period of operation time.

The co-presence of relatively high concentrations of CH_4 and O_2 can be observed in elsewhere when air addition starts and ends although an aerobic landfill gas maintained low CH_4 and O_2 concentrations during a period of landfill operation (Read et al., 2001; Lee et al., 2002 and Kim, 2005). Although air addition into an anaerobic landfill stimulates the growth of aerobic bacteria, the initial population of aerobes may not be enough to consume entire oxygen at once. Furthermore, high concentration of methane migrated from anaerobic zones may mix with air before oxidization, and it results in co-presence of high concentrations of CH_4 and O_2 becoming explosive for a short period. Oxygen concentration subsequently decreases in landfill gas as population of aerobes increases. A portion of migrated CH_4 is oxidized and converted into CO_2 by methane-oxidizer, and the rest of CH_4 is diluted with nitrogen and is discharged through the gas collection system. As shown in Figure 4.2, explosive region is smaller as O_2 is converted into CO_2. Collectively, landfill gas is less explosive as CO_2 concentration increases.

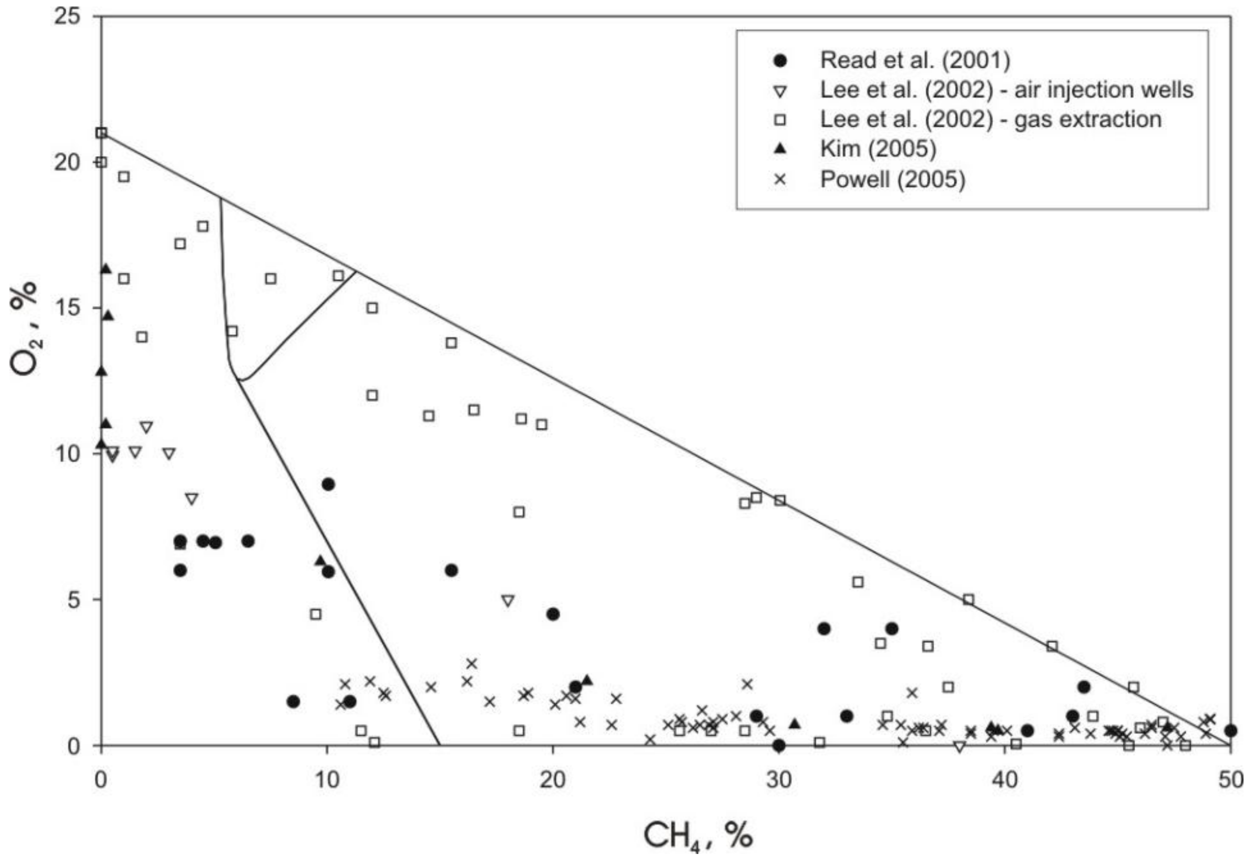


Figure 7.21. The distribution of LFG over an explosivity chart.

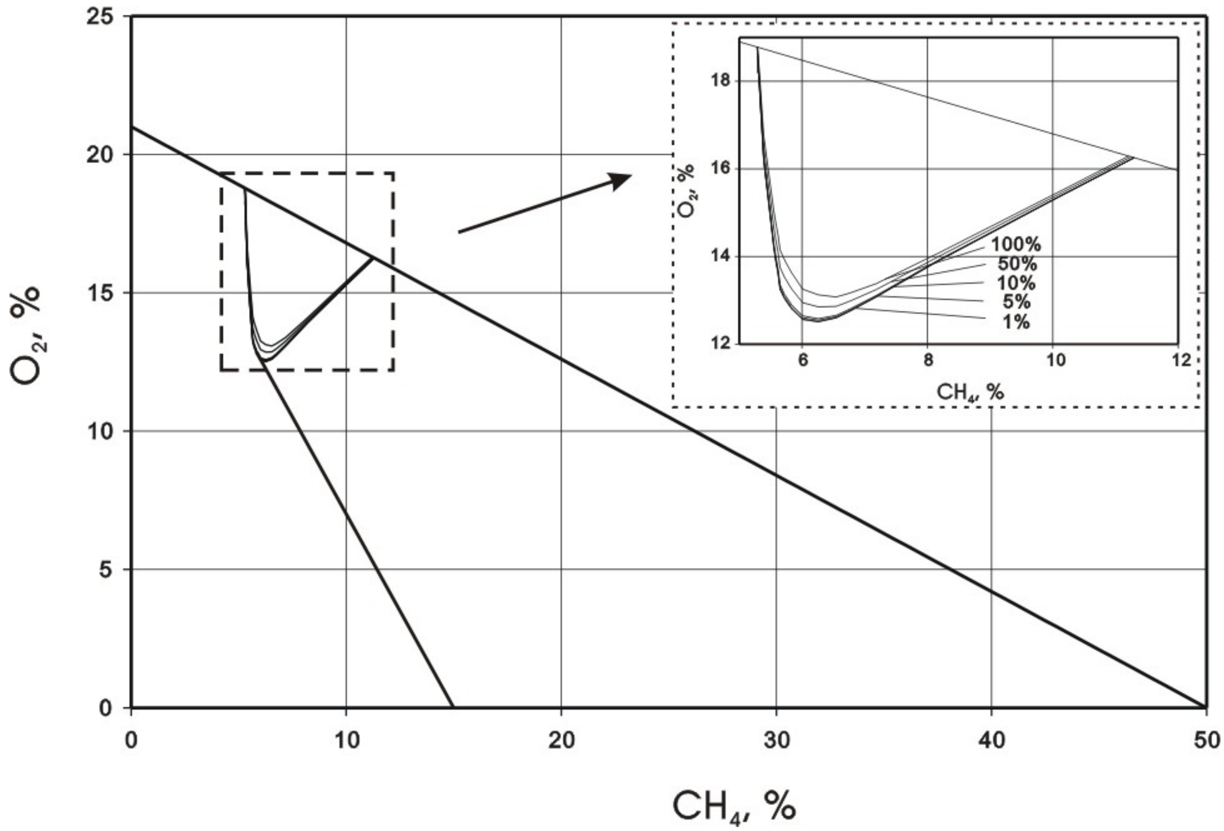


Figure 7.22. Explosivity Chart for Case Scenarios 2 and 3. The percentages in a box indicate conversion ratio of oxygen into carbon dioxide.

Appendix D. Peer-Reviewed Journal Articles

- Timothy G. Townsend, Hwidong Kim and Valerie Bonilla, “Flammability of Landfill Gas Mixtures: Considerations for Landfill Air Addition” (draft)
- Powell Jon (2005), “Trace gas quality, temperature control and extent of influence from air addition at a bioreactor landfill”, Master’s thesis, University of Florida

7.8 Estimation of Hydraulic Conductivity

This chapter reports the results of research conducted at a full-scale landfill to estimate the field saturated hydraulic conductivity of the landfilled MSW using the borehole permeameter technique. The site was equipped with a large number of vertical wells (as it was designed as a bioreactor); while a few wells did contain some initial standing water, the majority did not. Unlike other sites where in-situ waste hydraulic conductivity measurements have been reported, this site contained relatively new and undegraded waste; the data should thus prove valuable for design of liquids addition systems at bioreactors. The primary objective was to apply to the borehole method and to estimate the in-situ hydraulic conductivity; the impact of waste depth was explored. A secondary objective was to compare the permeability values (calculated from hydraulic conductivity) of the landfilled waste to previously reported values for the same well locations where air was used as the test fluid (Jain et al. 2005).

7.8.1 Borehole tests

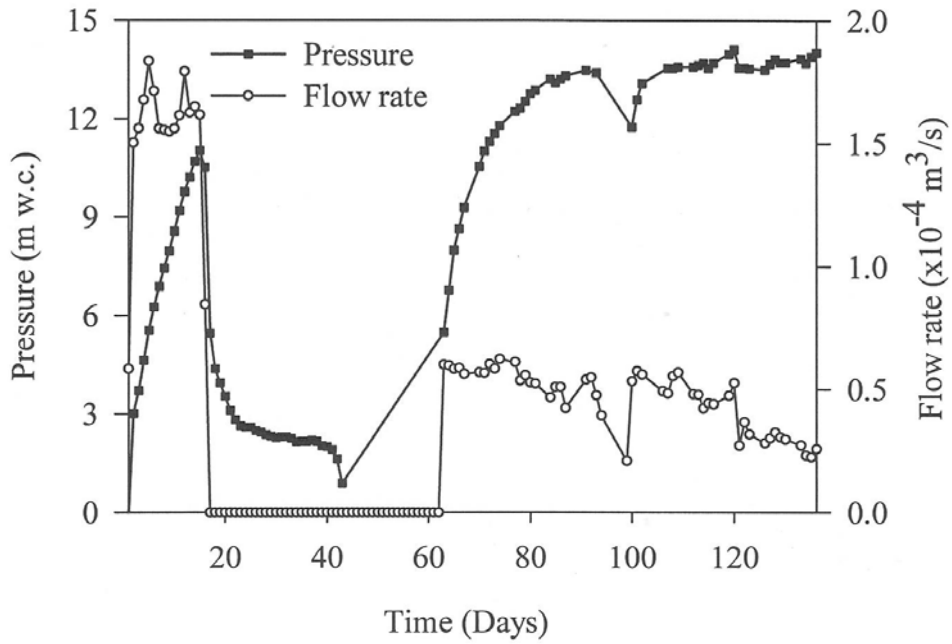
Liquid addition to the vertical injection wells began at the site in June 2003. For the most part, the liquids added consisted of leachate from the landfill cell where the experiment was conducted and from adjacent landfill units. On several occasions, when leachate production was low, the leachate storage basins were supplemented with groundwater. The borehole permeameter tests were conducted from March to October 2004 using 77 wells in 26 clusters. Liquid was added in the deepest wells first, followed by the middle wells and then the shallow wells. The moisture addition system was operated continuously except when the system was stopped for changing liquid injection locations, cleaning the in-line sediment filters, and during periods of severe inclement weather. Liquid was added to the wells at a constant rate until the water level or pressure stabilized; the flow rate added and the pressure at the bottom of wells were continuously monitored. The initial target liquid addition rates ranged from 3.2×10^{-5} to 1.9×10-4 m^3/s (1.9 to 11.4 L/min); these flow rates were reduced as necessary to maintain the liquid level in the well below the landfill surface to minimize the potential for surface seeps. In some cases, flow rates had to be adjusted down several times prior to reaching an approximate steady-state condition.

Liquid addition rates and water levels were measured using two different methods. At any given time, 12 injection wells were monitored continuously using single-jet impeller meters (ABB, USA) and submersible pressure transducers (GE Druck, Inc.). These devices provided a digital output which was recorded at a frequency of once per minute using a Campbell Scientific (CR10X) datalogger. In wells that were not equipped with continuous recording equipment, the liquid level was measured by lowering a pressure transducer into the well, and the liquid addition rate was measured using an impeller water meter equipped with a flow totalizer. If the liquid injection rate into a well was below the detection level of the flow meter, the flow rate was measured once or twice per week by recording the flow rate into a graduated cylinder. The water levels in the wells adjacent to the well being used for liquid addition were also monitored on occasion.

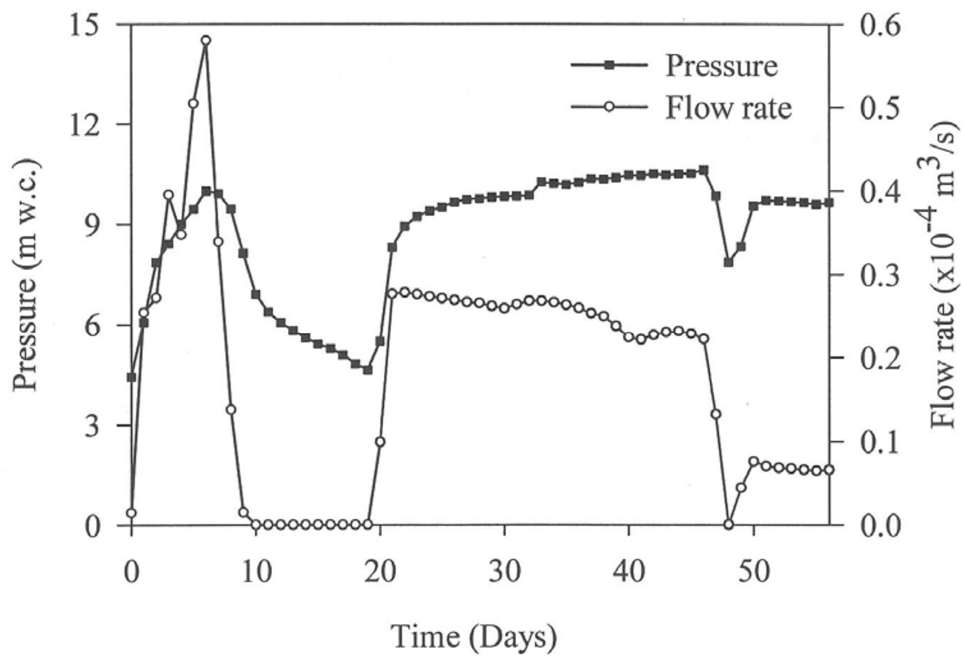
7.8.2 Hydraulic conductivity determination from field test

A total of 14,600 m^3 of moisture (11,200 m^3 of leachate and 3,400 m^3 of groundwater) was added during the borehole permeameter tests. During the initial operation of the liquid addition system, liquid levels in some wells reached elevations above the surface of the landfill. This resulted in liquids seeps at the surface of the landfill. Liquid addition in subsequent tests was carried out under conditions where the liquid level in the wells was maintained below the surface elevation of the landfill. As mentioned earlier, flow rates used at the beginning of a borehole permeameter test for a particular well ranged from 3.2×10-5 to 1.9×10-4 m^3/s (1.9 to

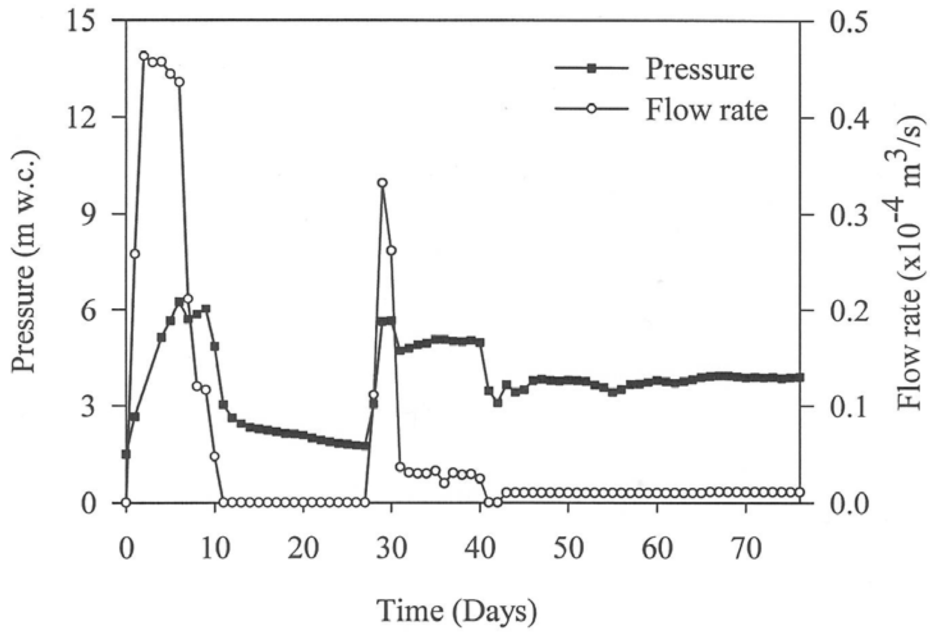
11.4 L/min); these flow rates were reduced as necessary to maintain the liquid levels within the landfill.



(a)



(b)



(c)

Figure 7. 23. Typical temporal variation of moisture addition rate and pressure at the bottom of well during borehole permeameter tests (a) deep well, (b) middle well and (c) shallow well

Figure 7.23 presents examples of typical borehole permeameter results. The liquid flow rate added to the well and the pressure head measured at the bottom of a well are plotted as a function of time for three individual wells. One plot each is provided for a shallow, middle and deep well at steady-state conditions; wells from the same cluster were not used in Figure 7-# as none of the well clusters had more than one well which reached steady-state at the end of the test. Although borehole permeameter tests were performed on 77 individual wells, only 38 wells were found to reach steady-state ever after periods of prolonged injection (up to 10 weeks in some cases). Upon examination of the data, it was discovered that many of the wells that had water levels near the surface of the landfill experienced short-circuiting of flow along some preferential path. For these wells, when the flow rate was reduced in an effort to lower the liquid level, the liquid level remained relatively constant. The preferential paths were hypothesized to result from the proximity of the wells to gravel-filled gas collection trenches on the surface, side slopes, and the surface itself. This phenomenon is illustrated in Figure 7.23 (a). At the end of the trial as the liquid addition rate was decreased by approximately one-half (at day 120), the liquid level in the well did not respond in a like fashion. Data from borehole tests where the liquids levels were within 1.5 m of the surface of the landfill were thus excluded from further Ks determination.

Values for Ks were estimated for the 23 wells which had reached conditions of steady-state and water levels at least 1.5 m below the surface of the landfill. Figures 7.23(b) and 7.23 (c) illustrate measured data for two wells that were used for the estimation of Ks. In these wells, a change in flow rate resulted in a response in liquid level in the well. For each of the days that a well was estimated to be at steady state, the average values of Q and H for that day were used to calculate Ks. The average Ks for a particular well location was then determined as the arithmetic mean of the Ks values. Only two of the wells, where Ks were estimated, had an existing water column (of approximately 2 m) prior to start of borehole test. As mentioned earlier, the presence of standing water does not conclusive indicate that a continuous phreatic water surface exists within the waste; it just suggests that the waste in the vicinity of the well is at higher percent saturation than that of waste surrounding wells with no initial water column. The borehole test results are independent of percent saturation of unsaturated media. Therefore, Ks for these well locations were also calculated using the same procedure as described above.

Table 7.4 presents the arithmetic mean, standard deviation, minimum, maximum, and 95% confidence interval of the Ks values corresponding to the well locations at the three different waste depths. The number of well locations where Ks was determined for each waste depth is also provided. As a reminder, Zangar's (1953) equation, which was used to calculate Ks in this study, is based on the assumption that the medium is isotropic whereas landfilled waste is commonly treated as an anisotropic medium, therefore, the estimated Ks are a combination of Ks in the horizontal and vertical direction. The Ks was found to range from 5.4×10^{-6} cm/sec to 6.1×10^{-5} cm/sec. The values of Ks were found to be on the lower end of the range of those reported previously (Ettala 1987, Oweis et al. 1990, Shank 1993, Townsend et al. 1995, Landva et al. 1998, Wysocki et al. 2003). The differences in the hydraulic conductivity measured and the previous studies could result from differences in waste characteristics, cover soil characteristics, differences in the landfill operations, and the different methods employed for determination of hydraulic conductivity. Many of the other studies were conducted on older landfills or were simulated in the laboratory, and these conditions may not be truly representative of new, well-compacted waste undergoing waste decomposition and active biogas production. Townsend et al. (1995) measured Ks using surface infiltration ponds on top of new, well-compacted waste and found values in the same range as reported here.

Table 7.4. Hydraulic conductivity values and its distribution

	Number of locations	Field saturated hydraulic conductivity, Ks (x 10^{-5} cm/sec)			
		AM +/- SM	Min.	Max.	95% confidence interval
Upper layer (3-6 m deep)	8	2.41 +/- 1.89	0.54	6.11	0.83 – 3.99
Middle layer (6-12 m deep)	10	1.22 +/- 0.58	0.56	2.34	0.81 – 1.64
Deep layer (12-18 m deep)	5	1.19 +/- 0.50	0.74	1.90	0.57 – 1.80

7.8.3 Conclusions

The hydraulic conductivity was estimated at 23 locations using the borehole permeameter test. The Ks of landfilled waste was found to range from 5.4×10^{-6} cm/sec to 6.1×10^{-5} cm/sec.

These estimates were in the low range of those previously reported in the literature, which is believed to be a reflection of the relatively new and well-compacted waste, and the impediment to liquid flow created by the presence of an entrapped gas phase. The mean Ks of waste in the shallow layer was found to be significantly greater than that of waste in the middle and deep layer. The estimated Ks should be interpreted as a combination of Ks in the vertical and horizontal directions. Waste permeability values (kw) were calculated based on the estimated Ks and compared to permeability values estimated using air as fluid (air permeability, ka). The ratio of permeability estimated using air as fluid to that using water as fluid (ka/kw) was found to range from approximately 220 to 3500; this difference was again primarily attributed to the presence of the entrapped gas phase. Other factors such as clogging of waste media (with biological growth and suspended solid present in the added moisture) and potential short-circuiting of air along the solid section of wells in the air pump tests may also have contributed to the difference between ka and kw. Landfill designers and modelers should use caution when applying Ks values from ka measurements, and vice versa.

Appendix D. Peer-Reviewed Journal Articles

- Jain, P., Powell, J., Townsend, T., Reinhart, D.(2006). "Estimating the hydraulic conductivity of landfilled municipal solid waste using the borehole permeameter test." Journal of Environmental Engineering, ASCE, 132(6), 645-652.

7.9 Conclusions and Summary

In this volume, the data collected from the NRRL, bioreactor, including waste decomposition, moisture content, settlement, leachate quality and concerns on air addition were used to assess the success of the project. The following major observations can be stated; (1) undecomposed waste was well degraded within the 6 years, research period, (2) the moisture content was substantially increased, and (3) great settlement was observed along with leachate addition. Additional conclusions from the different experiments are summarized as follows:

- In this research, the effects of applying leachate recirculation technique on waste decomposition were evaluated through moisture contents and BMP assay. Solid samples were excavated from NRRL landfill before and after moisture addition and characterized for waste composition, moisture content and methane potential. As a result of moisture addition for last 6 years, moisture content of the waste was substantially increased.
- Moisture contents of waste samples collected in 2001 did not vary by depth. Comparing methane yields between waste samples collected from top (0-10 ft) and bottom (40-50 ft), no significant differences were found from them. This would a typical 'dry-tomb' traditional sanitary landfill. It would not be expected to see great enhancement of waste decomposition for a few years. On the contrary, methane yields of solid waste collected in 2007 were substantially lower.

- Another analytical results show that waste decomposition is strongly associated with moisture content. However, the effect of moisture contents on waste decomposition was not substantially critical when moisture content were greater than 40%. These results indicate that excessive moisture addition may not substantially enhance waste decomposition. This research suggest that moisture would be necessarily added to a landfill for the purpose of keeping waste wet condition, mass (e.g., nutrient and microorganisms) transfer and leachate treatment.
- The waste decomposition rate of the New River Regional bioreactor landfill was estimated. BMP assay results of waste samples excavated in 2001 and 2007 were used; 2001 sampling day was set as an initial time, t_0 and M_{BVS} of 2001 samples were determined as $M_{BVS,0}$ and M_{BVS} of 2007 samples were used for $M_{BVS,\,t}$. Waste decomposition rate, k, calculated from these values was 0.14 year^{-1}. Waste decomposition rate estimated from the aerobic and anaerobic bioreactor landfills study using lab-scale lysimeters were 0.326, 0.368 and 0.119 and 0.162 year^{-1} for aerobic and anaerobic bioreactor landfills, respectively.
- Substantial settlement was observed in NRRL for last 7 years. GPS survey results indicate that the settlement occurred along with leachate recirculation. When comparing aerobic and anaerobic landfill bioreactors with respect of total settlement during period of investigation, greater settlement was observed in the aerobic lysimeters with approximately three times greater waste decomposition rate.
- The hydraulic conductivity was estimated at 23 locations using the borehole permeameter test. The Ks of landfilled waste was found to range from 5.4×10^{-6} cm/sec to 6.1×10^{-5} cm/sec. These estimates were in the low range of those previously reported in the literature, which is believed to be a reflection of the relatively new and well-compacted waste, and the impediment to liquid flow created by the presence of an entrapped gas phase. The mean Ks of waste in the shallow layer was found to be significantly greater than that of waste in the middle and deep layer. The estimated Ks should be interpreted as a combination of Ks in the vertical and horizontal directions.