Effects of Adding Wash Tower Effluent to Ano Liossia Landfill to Enhance Bioreaction

by
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ABSTRACT

A theoretical study was performed on the effects of adding sulfate-rich wash tower effluent from the Athens hospital waste incinerator to the Ano Liossia landfill of Athens. The method of mass balance was used to examine the production of leachate, the generation of methane, and the reduction of sulfates into sulfides. The water mass balance was performed using the method of Thornthwaite and the result indicated that the leachate collection facility at Ano Liossia landfill would be able to handle the additional leachate. The hydrocarbon-methane mass balance was performed using the EPA's LandGEM model which is based on first-order decomposition of the waste. A 26% difference between the generation of methane in a conventional landfill and a bioreactor landfill was predicted. Finally, a first-order model was developed by analogy to the LandGEM model to study the reduction reaction of sulfates into sulfides. The amount of hydrogen sulfide produced from solid waste disposed in the landfill dominated the amount of hydrogen sulfide produced from the additional wastewater.

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Tables and Figures

1.0 Introduction

The disposal of solid wastes is an increasingly important global issue due to the escalating growth in world population and the resulting increase in waste production. Of the available management options for solid waste disposal, landfilling is the most commonly employed method worldwide (Reinhart, 2002); the landfill is an engineered land method of disposing municipal refuse, industrial or agricultural residues, wastewater sludge, incinerator ash, recycle discards, and/or treated hazardous wastes in an environmentally safe way. It is an active system whereby biological, chemical and physical processes promote the degradation of wastes and the production of contaminated leachate and gas. The landfill design includes environmental barriers, such as liners and caps which exclude moisture (dry tomb landfill). Moisture, however, is essential to waste degradation. Waste stabilization can be accelerated if the landfill is operated as a bioreactor, instead of a dry waste isolation cell. This entails addition of moisture in the fill in a controlled manner (e.g. recirculation of leachate, addition of wastewater). Moisture recirculation increases the rate of establishment of anaerobic microbial population within the fill. As a result, the emitted loads of Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD), chlorides and metals are lowered, more landfill gas is produced, the post-closure monitoring period is reduced and waste settlement is enhanced leading to a 20% reduction in landfill heights (EPA Workshop on Bioreactor Landfills, 2000).

The Solid Waste Association of North America has defined the bioreactor landfill as a sanitary landfill operated for the purpose of transforming and stabilizing the readily and moderately decomposable organic waste constituents within 5 to 10 years following closure by purposeful control to enhance microbiological processes (Reinhart, 2002). The bioreactor landfill significantly increases the extent of waste decomposition, conversion rates and process effectiveness over what would otherwise occur within the landfill.

Solid waste disposal in Greece follows the global trend, since landfilling dominates the other disposal options. The Ano Liossia landfill is the main landfill of Athens; it serves 4 million people, and is scheduled to close in 2007. Given the lack of space in the densely-populated city of Athens,

the concept of extending the landfill's lifetime through bioreaction is particularly appealing. Landfilling in Greece is not a sustainable waste management option, since the available sites for landfill development are limited. Moreover, the very low charges for land disposal (approximately 20 - 25 $/tonne) reduce the financial feasibility of other options such as recycling and composting (www.compostnetwork.info/countries/greece.htm, url accessed 5/8/2006). Bioreactor landfills, therefore, could present a temporary solution to the lack of space problem until alternative disposal options become economical.

In this study, we wish to explore the effects of adding wastewater to the landfill in a controlled manner to enhance bioreaction. The source of the wastewater is the wash tower effluent from a nearby hospital waste incinerator. We will use the method of mass balance to address how three important landfill parameters change under bioreactor conditions: the amount of leachate produced, the amount of methane generated and the reduction of sulfates into sulfides.

2.0 Biological degradation of solid wastes

The stabilization of municipal solid waste occurs in two major phases:

1) An aerobic degradation phase, which happens almost immediately after waste placement and lasts a few hours to one week,
2) An anaerobic degradation phase, which develops once the oxygen originally present in the landfill is consumed and represents most of the landfill life.

Biodegradation proceeds readily in landfill environments due to the waste composition. According to Miller (2003), 63.1% of the refuse is comprised of cellulose and hemicellulose which start to biodegrade immediately.

2.1 Aerobic Degradation

During this phase, bacteria start growing on the surface of wastes and start metabolizing the waste by hydrolyzing complex organic structures to simple, soluble molecules (ITRC, 2006). Cellulose, hemicellulose, and proteins are converted to soluble sugars and amino acids. Leachate produced during the aerobic phase is also due to the dissolution of highly soluble salts initially present in the landfill and is most likely a result of moisture that was squeezed out of the wastes during compaction and landfill filling operations. The short duration of the aerobic phase is due to the high biochemical oxygen demand (BOD) of the solid wastes and the limited amount of oxygen present in a sanitary landfill.

2.2 Anaerobic Acid Production

When all the oxygen is depleted, the microorganisms cannot completely metabolize the soluble sugars and amino acids (ITRC, 2006). They begin to break them down to organic acids which are readily soluble in water and begin to accumulate in the landfill. As a result, a low pH leachate forms, as well as considerable concentrations of inorganic ions (e.g., Cl, SO_4, Ca, Mg, Na). The microorganisms involved in these processes obtain energy for growth from the chemical reactions that occur during metabolism and a portion of the organic waste is converted into cellular or exocellular material.

2.3 Methanogenic Degradation

The anaerobic conditions and the soluble organic acids formed during the first stage of anaerobic degradation create an environment where methanogenic bacteria can grow. Methanogenic bacteria utilize the organic end products from the first stage of anaerobic degradation and convert them into methane and carbon dioxide. Methane fermentation generally begins within one year following solid waste placement (ITRC, 2006). The methanogenic bacteria prefer a relatively neutral pH (6.6 to 7.4). Therefore, if acid formation in the first stage is excessive, i.e. pH is below 6, the activity of the methanogenic bacteria could be inhibited.

Having analyzed the stages of waste biodegradation, we can now address the concept of a bioreactor landfill since it is directly associated with how bacteria degrade the organic matter. We will present the types of bioreactors and their related consequences and then analyze the Ano Liossia case.

3.0 Types of Bioreactors

A bioreactor is meant to accelerate the decomposition of municipal solid wastes by distributing moisture, nutrients, enzymes, and bacteria throughout the waste mass more efficiently than natural infiltration alone.

3.1 Anaerobic bioreactor: The traditional method of landfill bioreactor operation involves enhancing waste stabilization by anaerobic microorganisms. Leachate and often additional water is recirculated into the waste mass in a controlled manner (ITRC, 2006). No air is injected into the landfill, since anaerobic conditions need to be attained for anaerobic bacteria to biodegrade the waste mass. Landfill gas is produced as a result of the biodegradation which can be captured to minimize greenhouse gas emissions and to produce energy.

3.2 Aerobic bioreactor: Recently, however, interest has been focused on the introduction of oxygen to the landfill. Leachate and often additional water is re-circulated into the waste mass in a controlled manner (ITRC, 2006). Air is simultaneously injected into the landfill through vertical or horizontal wells to enhance aerobic bacterial activity and accelerate waste degradation. The advantages of this type are accelerated biological activity, limited production of organic acids and the delay of methanogenesis. Concerns associated with this type of bioreactor include the fire hazards associated with injection of air and the high operating costs.

An area of further exploration is the use of combined anaerobic and aerobic systems. If such systems can be efficiently controlled, then they could offer a number of benefits. For example, the addition of air following anaerobic degradation would remove excess moisture from the landfill and fully compost the waste.

3.3 Hybrid bioreactor: a sequential aerobic-anaerobic treatment is employed to quickly degrade organic matter in the upper sections of the landfill, and collect gas from the lower sections (ITRC, 2006). The result is an early onset of methanogenesis.

3.4 Effects of a bioreactor

In this section we wish to explore the key advantages of a bioreactor landfill and justify why it would be worth operating the Ano Liossia conventional landfill as a bioreactor. These key advantages include:

- Efficient utilization of permitted landfill capacity
- Stabilization of waste in a shorter time and therefore reduced post-closure care
- Increased production of landfill gas and therefore increased revenue stream

Decomposition and biological stabilization of the waste in a bioreactor landfill can occur in a much shorter time frame (years) than what occurs in a traditional dry tomb landfill (decades). As a result, the density of the waste increases and the volume of the landfill can be reduced by as much as 20% (EPA Workshop on Bioreactor Landfills, 2000). If the resulting volume reduction of the waste can be reclaimed, then the lifetime of the landfill can be extended, which entails increased revenues. A secondary benefit from the efficient use of existing landfill capacity is the need for fewer landfills placed in green spaces.

As degradation of waste proceeds, the waste becomes less and less of a threat to human health. The waste pile becomes more stable as the density rises and the leachate quality in a bioreactor can improve with time. As a result, the amount of post-closure care can be reduced. Currently, the standard post-closure care period in the US is 30 years; however, should the bioreactor concept prove to be successful, these regulatory barriers may ease. If the overall length of the post-closure care period cannot be reduced, it is still possible that individual aspects of post-closure care can be re-evaluated and reduced.

Landfill gas (LFG) is generated earlier in the landfill operation for bioreactors than for conventional landfills (Figure 1). As a result, operational advantages associated with the generation of LFG can be realized sooner. These include direct income associated with the use or sale of the gas and the indirect advantage of increased LFG generation early in landfill operation. Diminishing gas generation late in landfill closure and post-closure provides another basis for reducing the post-

closure care period. Certainly, early incorporation of LFG collection and management system are important parts of bioreactor landfill design and construction.

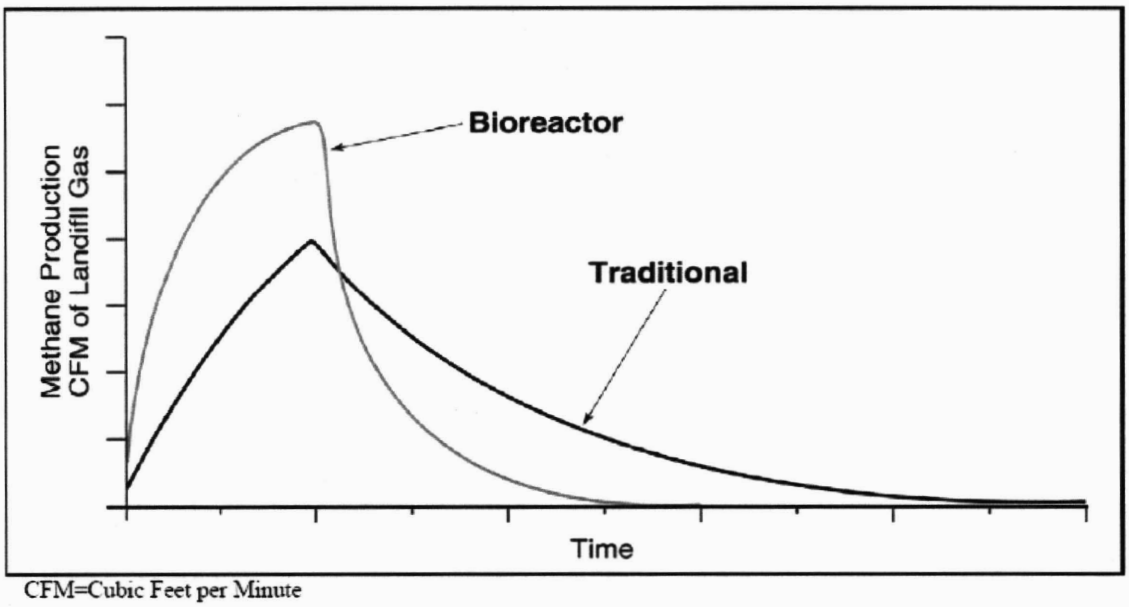

Figure 1: LFG generation

(ITRC, 2006)

3.5 Regulatory Status

The concept of a bioreactor landfill is not practiced in Greek landfill engineering because current regulations prohibit the recirculation of leachate without treatment (114218 ΦΕΚ Β' 1016/17.11.97). However, we hereby wish to explore this issue on a small scale in order to evaluate the advantages of bioreactor landfill and to acknowledge any environmental burdens.

In the United States, the bioreactor concept is relatively new, since it was not until October 9, 1991 that the United States Environmental Protection Agency issued minimum national standards for municipal landfills under the provisions of 40 CFR Part 258 of RCRA Subtitle D. The regulations of Subtitle D allow leachate recirculation, provided a composite liner and leachate collection system are included in the design (EPA Workshop on Bioreactor Landfills, 2000). As the US model is often mimicked by other countries worldwide, the regulation for bioreactor landfills in Greece may eventually follow EPA's example.

4.0 Ano Liossia Landfill

The Ano Liossia Landfill (Figure 2) receives 6,000 tonnes of municipal solid waste each day, serving 95% of the entire Attika region. At present, the landfill in operation consists of 6 cells, with total surface area of 275,000 m^2 (68.8 acres). The landfill began operations in 2003, and is expected to close in 2007.

Figure 2: Ano Liossia Landfill

4.1 Liner system

The liner construction of the landfill consists of the following (from lowest upwards):

- **Clayey soil** with thickness equal to 50 cm and hydraulic conductivity of $k = 5 \times 10^{-10}$ m/sec
- **Geosynthetic membrane** of high density polyethylene (HDPE) with thickness equal to 2.0 mm.
- **Geotextile** for membrane protection, weight 600 g/m^2.

- **Fine Sand layer** of thickness equal to 10 cm.
- **Drainage layer** consisting of sand and gravel. Thickness equal to 50 cm.

4.2 Leachate collection and treatment

Leachate is collected via a piping network from the entire landfill area and is sent to a leachate processing facility, where it goes through the following stages: sedimentation, anaerobic biological treatment, aeration, chemical precipitation, sand filtration, and activated carbon filtration.

The end product (see Figure 3) is stored in a tank in order to be used as irrigation water for the green space to be developed following landfill closure. Part of the final product is temporarily sent back to the landfill via a pipe distribution system, until a system of evaporators is put in place at the site. The amount of treated leachate that is recirculated is calculated so as to maintain the water balance and enhance the rate of biodegradation. The water is being discharged into a 1-m thick layer of gravel, where pipes distribute it into a wide area within the fill (instead of a point) in order to avoid channelling and ponding within the waste mass.

Figure 3: Final Product of Leachate Treatment System

4.3 Ano Liossia Landfill as a bioreactor

Having introduced the landfill of interest, we can now address the possibility of operating this landfill as a bioreactor. As mentioned in Section 3.5, untreated leachate recirculation is prohibited in Greece. Therefore, we will explore the issue of recharging wash tower effluent, instead of leachate, into the landfill. The effluent is generated at the hospital waste incinerator of Athens, which is conveniently located in the area surrounding the landfill (distance from landfill is approximately 1 km, see Figure 4).

Figure 4: Path of wash-tower effluent

The daily flow of water from the incinerator is 6 m^3. Since this flow is not sufficient to provide moisture for the entire landfill, we will examine a small area within the landfill, and draw our conclusions from a small-scale study. In September 2000, EPA held a workshop on bioreactor landfills and one of the conclusions reached by the participants was that in order for a landfill to be considered a bioreactor, it needs to receive 0.14 m^3/tonne of water per metric ton of waste (EPA Workshop, 2000). To evaluate the area of interest for our small-scale study, we used this information, along with a density of waste of 0.8 tonne/m^3, height of each cell of 57 m (Data obtained from Ano Liossia Landfill), and a time frame of 4 years. Our calculations (see Appendix A) indicate that the area of interest is 1,416 m^2. Our mass balance calculations, therefore, will depend on an individual landfill cell of area 1,416 m^2.

4.4 Water mass balance

A major issue in operating a bioreactor landfill is the impact of liquid addition on the leachate generation rate, since the leachate held on the liner will increase. Therefore, we performed a mass balance on the water in order to evaluate the impact of adding an extra 6 m^3 of liquid per day to the stability of the landfill.

According to data from the leachate collection facility in Ano Liossia, the maximum amount of leachate that the existing leachate collection system can handle is **390 m^3/day**. For our given area, this is equivalent to 325 inches/month.

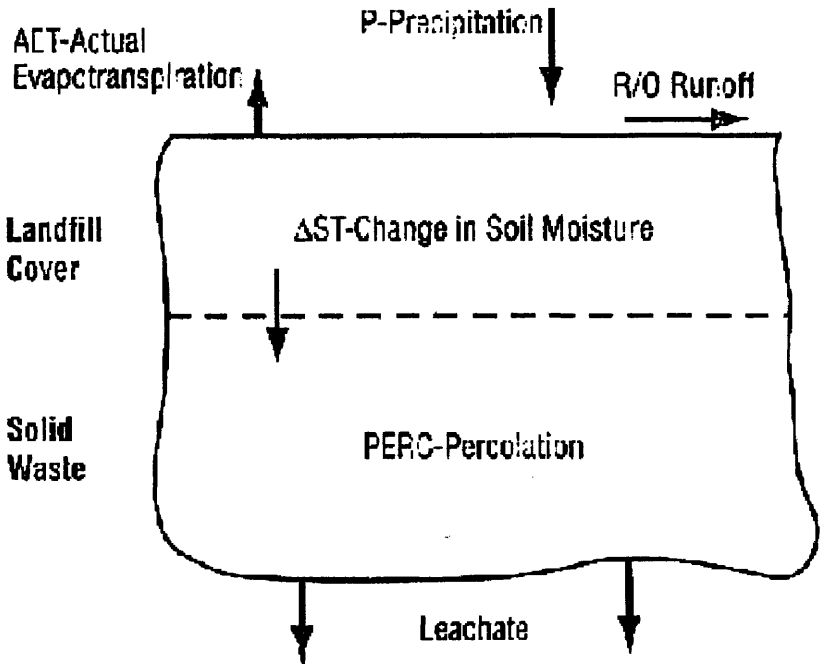

Figure 5: Water mass balance

(www.bvsde.ops-oms.org/muwww/fulltext/repind49/lesson4/lesson4.html)

Figure 5 indicates the water mass balance of a landfill. According to this diagram, the amount of leachate generated in a landfill is equal to:

$$PERC = P + R/O - ACT \quad (1)$$

where

PERC = amount of water percolated into the landfill, i.e. leachate [inches]

P = precipitation [inches]

R/O = surface runoff, equal to zero for an active cell [inches]

ACT = actual evapotranspiration [inches]

In order to evaluate *PERC* therefore, we need precipitation and evapotranspiration data for the region of Ano Liossia. Average precipitation and temperature data for each month were obtained from the weather stations of Elefsina, Nea Philadelphia and Tatoy (Table 1).

Table 1: Precipitation and Temperature data

Month	Average air temperature (°C)	Precipitation (mm)
January	8.1	54.3
February	8.7	47.8
March	10.9	42.9
April	15.0	27.2
May	20.2	21.1
June	25.1	10.6
July	27.4	7.4
August	27.0	4.7
September	23.0	13.7
October	17.8	49.6
November	13.4	57.4
December	9.8	69.2
Yearly average	**17.2**	**405.9**

There exist various formulas to evaluate the actual evapotranspiration (ACT term) in the water mass balance. The most widely used are the formulas by Turk (1955), Coutagne (1935) and Thornthwaite (1957). The first two provide evapotranspiration values based on yearly averages of precipitation and temperature. We chose Thornthwaite's method because it calculates evapotranspiration on a monthly basis, thereby taking into account monthly fluctuations of the climate parameters.

The Thornthwaite method is a tabular procedure to determine evapotranspiration. It was originally developed for natural soils, but was subsequently adapted to landfill analysis.

The modified mass balance equation we used to account for the wash tower effluent is the following:

$$P + IR = PERC + AET + \Delta ST \qquad (2)$$

where

IR = wash tower effluent [inches]

ΔST = change in storage [inches]

Table 2: Water balance calculations

	January
Air Temperature (°C)	8.1
Heat Index (table)	2.08
Precipitation (inches)	2.14
Surface Runoff	0
IR, wash tower wastewater input	5.01
I, infiltration (inches)	7.14
I – PET (inches)	6.64
AET, actual evapotranspiration (inches)	0.50
PERC (inches)	**6.64**

In Table 2 we have included the Thornthwaite method only for January (see Appendix B for entire mass balance). Our results indicate that the highest leachate generation occurs in January, when *PERC* is equal to 6.64 inches/month (i.e. leachate flow of **239 m^3/day)**. This is significantly below the maximum capacity of the leachate collection facility, which, as indicated above, is **390 m^3/day**. We can conclude, therefore, that the leachate collection system has sufficient capacity to handle the increased leachate generation due to bioreaction.

4.5 Hydrocarbon-Methane mass balance

Waste stabilization occurs by conversion of acetic acid into methane during the stage of methanogenesis discussed above. The two products are methane, which is essentially insoluble in water, and carbon dioxide, which either escapes as gas or is converted to bicarbonate alkalinity through its equilibrium relationship with $H_2CO_3(aq)$. The bacteria responsible for methanogenesis use formic acid, acetic acid, methanol and hydrogen as their energy sources. Of these, acetic acid and hydrogen serve as the major substrates for methane formation: approximately 72% of the methane formed in the anaerobic digestion of waste comes from acetate cleavage (Parkin and Owen, 1986), and the remaining 28% results from reduction of carbon dioxide:

$$\mathbf{CH_3COOH \rightarrow CH_4 + CO_2} \qquad (3)$$

$$CO_2 + H_2 \rightarrow CH_4 + H_2O \quad (4)$$

Anaerobic bioreactors generate landfill gas (principally methane and carbon dioxide) earlier in the waste degradation process, and at a much higher rate than the traditional landfill (ITRC, 2006). This is because the accelerated decomposition process depletes the source waste. Depending on how a particular bioreactor is operated, this early landfill gas production can be viewed either as an advantage or as a disadvantage. Benefits offered by the accelerated methane production in a bioreactor where landfill gas is captured are the availability of gas for productive uses and the fact that landfill gas (LFG) impacts to the atmosphere, groundwater, or to potential receptors are reduced.

Gas production rates at landfills vary significantly, depending on the waste types and moisture content of the wastes. Mathematical and computer models for predicting gas yields are widely available for landfill operators, energy recovery project owners, regulators and energy users who need to be able to project the volume of gas produced and recovered over time from a landfill (EPA 2005). To explore the influence of bioreaction on the methane generation rate of the Ano Liossia landfill, we have performed a hydrocarbon-methane mass balance on both the conventional landfill and the bioreactor landfill in order to compare their landfill gas generation rates.

The model we chose was EPA's Landfill Gas Emissions Model (LandGEM), a software application with a Microsoft Excel interface that estimates air pollutants and other gases from municipal solid waste landfills. LandGEM is based on the following first-order decomposition rate equation for quantifying emissions from the decomposition of landfilled waste:

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} kL_o \left(\frac{M_i}{10} \right) e^{-ktij} \quad (5)$$

where:

Q_{CH4} = annual methane generation in the year of the calculation (m^3/year)

i = 1-year time increment

n = (year of calculation) – (initial year of waste acceptance)

j = 0.1-year time increment

k = methane generation rate ($year^{-1}$)

L_0 = potential methane generation capacity (m^3/tonne)

M_i = mass of waste accepted in the i^{th} year (tonne)

t_{ij} = age of the i^{th} section of waste mass M_i accepted in the j^{th} year (decimal years, e.g., 3.2 years)

Inputs to the model included the following:

- Landfill open year (represents the year that the landfill began accepting waste) = **2003**
- Landfill closure year (represents the final year the landfill accepted waste) = **2007**
- Waste acceptance = **16,149 tonnes/year**
- L_o = **120 m^3/tonne**
- Conventional landfill: k = **0.05 $year^{-1}$**
- Bioreactor landfill: k = **0.3 $year^{-1}$**

The most important parameters to the model are the methane potential of the waste (L_o) and the methane generation rate (k). The value of L_o was obtained from the Ano Liossia landfill engineers and is based on local parameters. The k values were obtained from EPA's publication *First-Order Kinetic Gas Generation Model Parameters for Wet Landfills* (USEPA, 2005), based on climatic conditions in Attica region and are shown in Table 3. Greece falls under the "intermediate" climate, and hence an intermediate k is used.

Table 3: Methane generation rate, k, as a function of climate

k ($year^{-1}$)	Climate
0.1 - 0.35	Wet
0.02 - 0.1	Dry
0.05 - 0.15	Intermediate

(Data obtained from Ano Liossia Landfill)

The generation of methane is particularly important for the Ano Liossia landfill because a natural gas power plant (see Figure 6) has been in operation at the landfill since March, 2001.

Figure 6: Natural gas plant

The power plant is operated as follows: Methane is collected from the landfill through 243 vertical pumping wells which are connected to perforated pipes inside the waste bulk. Each well is connected via horizontal pipes to a substation. The gas is transported to the station by a horizontal pipe network (25,000-m long). All the biogas is concentrated at the substation where it goes through a first degree of processing. It is then transported via a second horizontal pipe network to each of the 11 electricity generation units. At the electricity generation unit, biogas is dehydrated and further processed before it ultimately goes into combustion to produce electricity (see Figure 7).

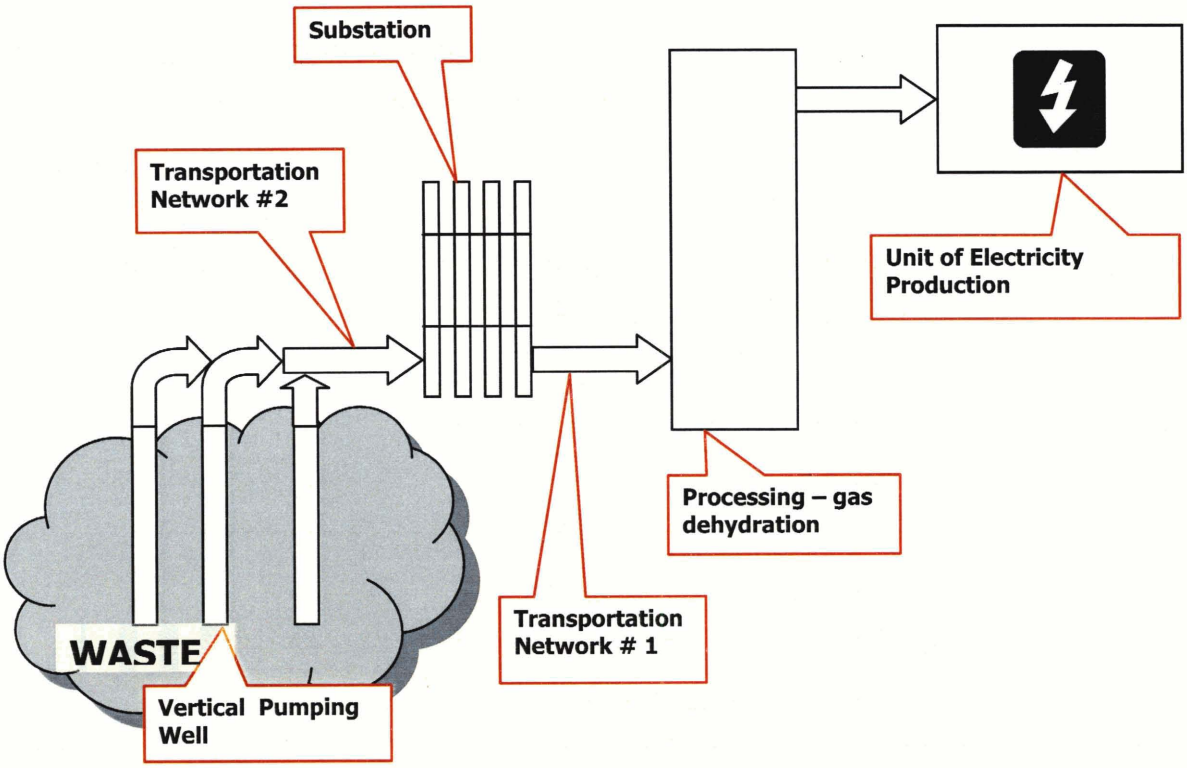

Figure 7: Electricity production from methane

(Data obtained from Ano Liossia Landfill)

The mass balance for a landfill cell in Ano Liossia in conventional and bioreactor operation yields the results shown in Figure 8. (For detailed calculations see Appendix C.)

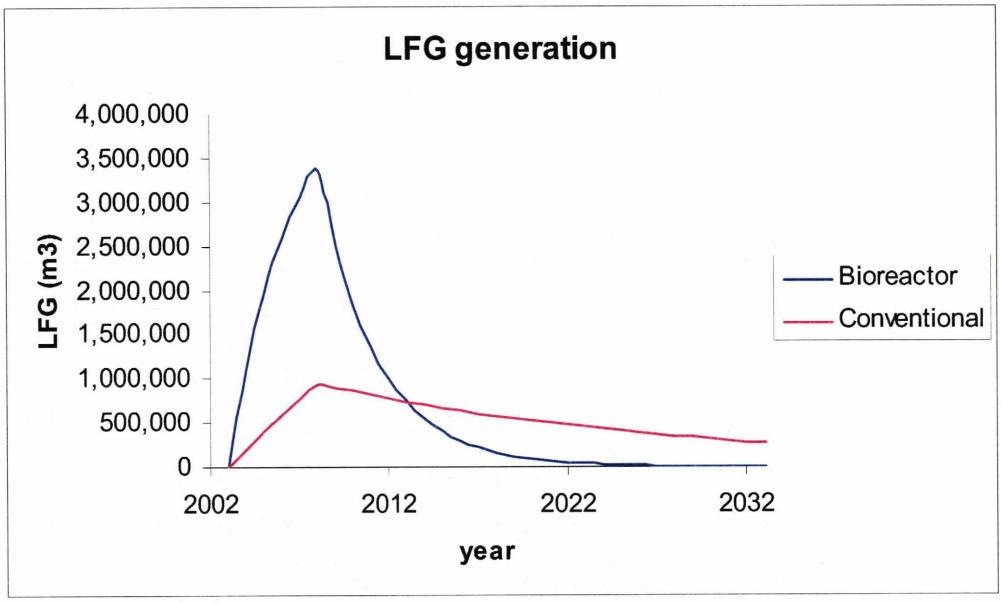

Figure 8: Predicted LFG generation at Ano Liossia Landfill

The curves in Figure 8 indicate the difference in the two rate constants, depending on whether the landfill is wet or dry. The difference in the total amount of LFG produced between the two landfills is 26%, as shown in Table 4, thereby making the bioreactor option appealing to methane power plants on landfill sites.

Table 4: Comparison of predicted LFG production at Ano Liossia Landfill

Total LFG produced from landfill (2003-2033)			
	Bioreactor	Conventional	% difference
LFG (million m^3)	**21.5**	**16.1**	26

4.6 Sulfates mass balance

In addition to methane and carbon dioxide, landfill gas also contains a number of trace components, such as hydrogen sulfide and volatile organic chemicals (Fairweather, 1998). Hydrogen sulfide is a colorless, flammable and highly toxic gas known for its rotten egg smell. It is produced by anaerobic

decay of the waste aided by sulfate-reducing bacteria, which use sulfate as the oxidizing agent according to equation 6 below.

Fairweather argues that in anaerobic ecosystems containing sulfate, sulfate-reducing bacteria and methane-producing bacteria are in competition for fermentation intermediates, such as hydrogen and acetate. The two electron sink processes are:

$$C_6H_{10}O_5 + 3SO_4^{2-} + 3H^+ \rightarrow 6CO_2 + 3HS^- + 5H_2O \quad (6)$$

$$C_6H_{10}O_5 + H_2O \rightarrow 3CO_2 + 3CH_4 \quad (7)$$

Fairweather's results indicate that for high sulfate and limiting acetate and hydrogen concentrations, methane production can be inhibited as the sulfate-reducing bacteria lower the hydrogen partial pressure to levels lower than the threshold concentration necessary for use by the methane-producing bacteria. Isa et al. (1986), however, argue that methane production can occur concurrently with sulfate reduction even under high sulfate concentration because there can exist sufficient organic matter to support both processes. Fairweather's study also indicates that substantially more cellulose is degraded through methanogenesis than through sulfate reduction, making the methane production the stoichiometrically more important electron sink process. If organic carbon is not limited, therefore, there is no inhibition between the two processes. In the case of a landfill, the organic matter is not limited, and thus we expect no interference in the methane production process due to sulfates.

Hydrogen sulfide production in landfills is a concern because of its toxic nature, its odor and its corrosive effects on landfill gas-energy equipment. Parkin and Owen (1986) have analyzed the effects of toxic substances in biological waste treatment and argue that methane-producing bacteria are more sensitive to toxicants than aerobic organisms. For sulfide in particular, Parkin and Owen report that a concentration of 200 mg/l is strongly inhibitory. However, they also emphasize that this concentration does not consider acclimation, which may significantly increase the concentration that can be tolerated by the bacteria. In fact, sulfide concentrations higher than 200 mg/l can be tolerated by methane-producing bacteria if a sufficiently large solids retention time is allowed for the bacteria

to acclimate to or metabolize the toxicant. In our case, the concentration of sulfate in the wastewater (975 mg/l), which under anaerobic conditions is expected to reduce to sulfide (whose concentration will be lower than 975 mg/l), is a lot higher than the reported concentration of sulfide (200 mg/l). However, the essentially infinite solids retention time at the landfill and the abundant organic matter are expected to minimize the toxicity effect.

Currently at Ano Liossia landfill, hydrogen sulfide production is not an issue because it represents only 0.0036% by volume of the total landfill gas (see Appendix D). However, in the bioreactor scenario we expect higher hydrogen sulfide concentrations. This is because the wash tower effluent from the hospital waste incinerator contains high concentrations of sulfate (975 mg/l), produced during the air pollution control stage of the incineration process. Table 5 shows the chemical composition of the wastewater of interest.

Table 5: Chemical composition of wash tower effluent

Parameter	Value	Unit
pH at 26°C	8.9	
Conductivity	119200	μS/cm
Alkalinity	1050	mg/l
Total Hardness	64.2	mg/l
Noncarbonate Hardness	0	mg/l
Carbonate Hardness	64.2	mg/l
Bicarbonates	3386	mg/l
Carbonates	1260	mg/l
Cl	35800	mg/l
Ammonia	3.2	mg/l
Nitrate	4.5	mg/l
Sox	975	mg/l
Ca	1140	mg/l
Mg	75	mg/l
K	95	mg/l
Na	24000	mg/l
Fe	228	mg/l
Mn	1.7	mg/l
Total Suspended Solids	2035	mg/l

In this section, we wish to predict the amount of hydrogen sulfide produced during bioreaction in Ano Liossia, and evaluate the significance of the result in terms of the viability of bioreaction with the use of wash tower effluent.

By analogy with the mass balance performed on methane production, we assume a first-order sulfate decomposition rate. Our model predicts that the amount of hydrogen sulfide produced in this bioreactor will be the sum of the amount of hydrogen sulfide produced from the sulfates contained in the waste mass plus the amount of hydrogen sulfide produced from the added sulfates from the wash tower effluent:

$$Q_{H_2S} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k' L'_o \left(\frac{M_i}{10}\right) e^{-k' t_{ij}} + \frac{k'}{\rho_{H_2S}} Q_{SO_4} e^{-k' t} \qquad (8)$$

where:

Q_{H2S} = annual hydrogen sulfide generation in the year of the calculation (m^3/year)

i = 1-year time increment

n = (year of calculation) – (initial year of waste acceptance)

j = 0.1-year time increment

k' = rate of reduction of sulfate into sulfide ($year^{-1}$)

L'_0 = potential hydrogen sulfide generation capacity (m^3/tonne)

M_i = mass of waste accepted in the i^{th} year (tonne)

t_{ij} = age of the i^{th} section of waste mass M_i accepted in the j^{th} year (decimal years, e.g., 3.2 years)

Q_{SO4} = annual incoming mass of sulfate from wash tower effluent (tonnes/year)

ρ_{H2S} = density of hydrogen sulfide (tonne/m^3)

Our model consists of two parts: the amount of H_2S produced by the waste, and the amount of H_2S produced by the added sulfates. The first part predicts the amount of hydrogen sulfide that will be produced in a bioreactor landfill according to the first order decomposition of hydrocarbons. L'_0 represents the potential of the waste to produce hydrogen sulfide and is estimated according to Miller's stoichiometric approach to anaerobic processes (Miller, 2003). He represents the waste mass

as the generic substance $C_aH_bO_cN_dS_e$ and concludes that the amount of hydrogen sulfide produced per mass of waste is equal to:

$$X_{H_2S} = \frac{34.076e}{12a + b + 16c + 14d + 32.06e} \tag{9}$$

To obtain the waste potential for hydrogen sulfide L'_0, we divide X_{H2S} by the density of hydrogen sulfide:

$$L'_o = \frac{34.076e}{12a + b + 16c + 14d + 32.06e} * \frac{1}{\rho_{H_2S}} \tag{10}$$

To obtain values for a, b, c, d and e we need the composition of the waste. According to Barlaz (1989), the composition of the refuse is on average 51.2 % cellulose and 11.9% hemicellulose. The chemical formulas of the two substances are known, and therefore a, b, and c are given according to the following chemical formula: $C_6H_{10}O_5$

We assume d to be zero for this calculation (i.e. no significant amount of nitrogen in the waste). To calculate the amount of sulfur in the waste, we use the following procedure:

The left column in Table 6 contains an approximate composition of the waste by volume, obtained from the operators of the Ano Liossia landfill. Due to lack of data with respect to density of each waste type, we have assumed that we can treat the waste fraction by volume as a waste fraction by mass. In the second column, we have matched each type of waste to the corresponding mass of sulfur per unit mass given by Miller (2003). The third column represents the mass of sulfur in the waste per unit mass and is a result of the multiplication of columns A and B.

Table 6: L'_0 calculation

A		B	C
Waste composition (by volume)		**Mass of Sulfur per unit mass (Miller, 2003)**	**Sulfur in waste**
Organics	0.465	0.190	0.0884
Paper	0.234	0.160	0.0375
Plastic	0.108	0.0300	0.00324
Metal	0.0374	0.0100	0.000374
Glass	0.0342	0.0300	0.00103
Cloth – leather – wood - rubber	0.0425	0.0200	0.000850
rest	0.0785		
		Total	**0.130**
		Moles of sulfur / mass of waste =	**0.00410**

The final row in column C of Table 6 is the result of dividing the total sulfur in waste (grams of sulfur/mass of waste) by the molecular weight of sulfur, in order to obtain moles of sulfur per mass of waste, which is equal to e in above equation.

Plugging in the values for a,b,c,d and e we obtain the following value for L′o:
0.000941 m^3 H_2S/tonne waste

The other input to the first part of our model is the hydrogen sulfide generation rate, k'. We assume that this rate is the same as the rate for the second part of our model, which represents the amount of hydrogen sulfide produced as a result of the sulfates in the wastewater. We base this assumption on the fact that both processes will be occurring at the same time, in the same place, and with the same bacteria. We assume that this process is also a first-order decomposition of sulfate into sulfide according to Equation 3. Inputs to this part of the model are:
Q_{SO4} = 2.14 tonnes/year
$k' = 0.049\ day^{-1}$

The flow of sulfates into the waste is a known value, since we know the flow rate of wash tower effluent (6 m^3/day) and the concentration of sulfates in the water (975 mg/l). The value of k′, however, is unknown and was obtained from literature review. The value of k′ represents the rate of the reduction of sulfate (reaction (2) from above). According to Rittman and McCarty (2001), under the assumption of anaerobic decomposition, k′ is approximately 0.049 day^{-1}. Pareek (1998) lists values for hydrolysis rate constants, but differentiates between SRB (Sulfate-Reducing Bacteria) dominating conditions, and MPB (Methane-Producing Bacteria) dominating conditions. For our case (MPB dominating), Pareek and Rittman and McCarty present the same value for k′.

Our model produces the results shown in Figure 9 (for detailed calculations see Appendix E).

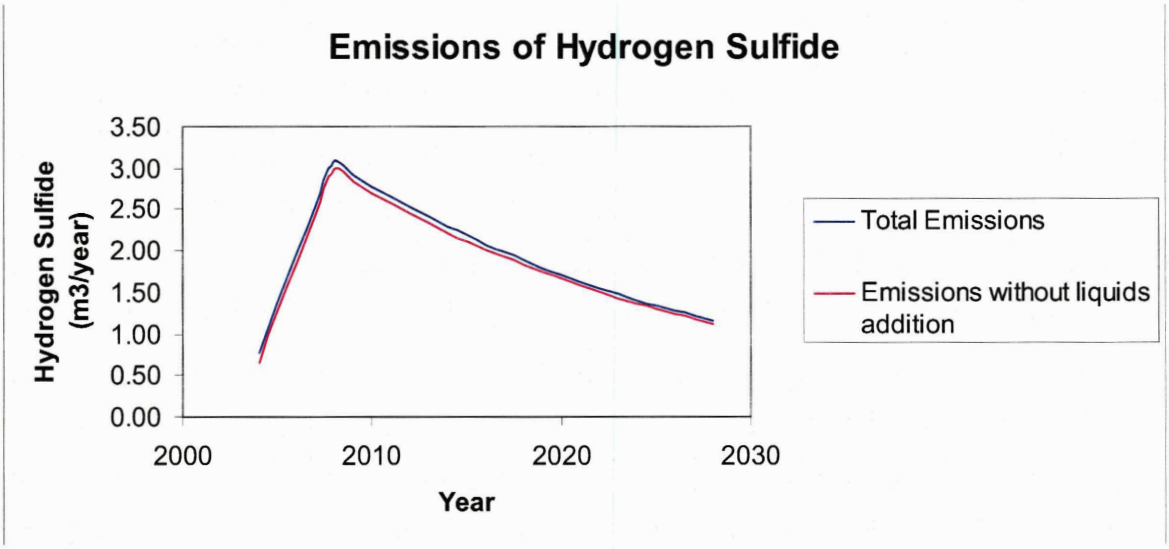

Figure 9: Hydrogen Sulfide production

The amount of hydrogen sulfide produced is dominated by the sulfates already in the waste. The amount of hydrogen sulfide produced due to the sulfates added represents 0.3% of the total emissions of H_2S.

This result is somewhat surprising since we expected that the addition of high sulfate wastewater into the landfill would significantly increase the production of hydrogen sulfide. The explanation for this outcome is that the mass flow rates from the two processes (sulfate reduction in waste and

sulfate reduction in wastewater) differ by an order of magnitude: the mass of sulfates in the waste is 14 tonnes/year, whereas the mass of sulfates in the wastewater is 2.1 tonnes/year. Therefore, the high sulfate concentration of the wastewater is not so high when expressed in terms of mass. According to EPA's requirement for bioreaction – 0.14 m^3 of water/tonne of waste (EPA Workshop, 2000) – 6 m^3/day is enough water to achieve bioreaction for a landfill cell of 16,149 tonnes/year, but as it turns out not enough to alter the decomposition reactions within the landfill.

Our results signify a hopeful statement for bioreactors in the future. The bioreaction application can be extended from just leachate recycling to other wastewater recycling from a variety of plants.

5.0 Consequences

5.1 Settlement

A significant consequence of bioreactor operation is enhanced settlement rates. Settlement is caused by the following factors (Reinhart, 1998):

- reduction in void space and compression of loose materials due to overburden weight,
- volume loss due to biological and chemical reactions,
- dissolution of waste matter by leachate,
- movement of smaller particles into larger voids, and
- settlement of underlying soils

Wet cell technology has been shown to enhance the rate and extent of subsidence. At the Sonoma County, California, experiments with pilot-scale landfills indicated that the recirculated cell settled by 20%, whereas dry cells settled less than 8% (Reinhart, 1998). As a result of rapid and somewhat predictable settlement, valuable air space can be utilized before closure of the cell. Therefore, waste filling procedures should homogenize the waste to the maximum extent practicable to help achieve field capacity throughout the landfill. Optimal operation includes subjecting all of the waste to re-circulated liquids and minimizing preferential flow paths. Figure 10 indicates the relationship between settlement rate and volume of added liquids (ITRC 2006).

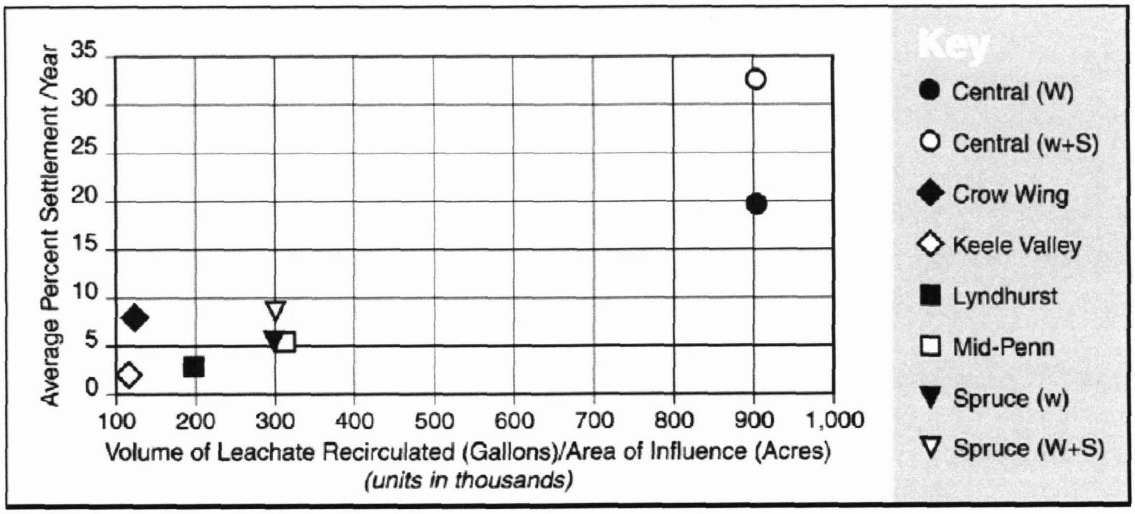

Figure 10: Average percent settlement/year vs. volume of leachate

Real data, therefore, indicate some correlation between the volume of leachate recirculated and the amount of waste settled.

5.2 Revenue from methane

As we discussed in the advantages of a bioreactor, the production of methane is significantly enhanced in a wet cell. For landfills that collect landfill gas, this increase can be translated into additional income due to electricity production from methane gas. Our model suggests that a wet cell would yield a 26% increase in methane production (see Table 4). As a result, we would expect a 26% increase in the revenue stream due to the increase in the number of kilowatt-hours produced annually.

6.0 Implementation

There are various technologies to introduce liquids into a bioreactor in a controlled fashion, each of which has significant advantages and limitations. Available technologies include: surface infiltration ponds, surface spraying, horizontal trenches, vertical wells, vertical injection needles, vertical injection wells, drip irrigation tubing, and area infiltration systems. The references for this section are Reinhart (2002) and ITRC (2006).

6.1 Surface and Surface Infiltration Ponds

This technology involves the construction of a shallow infiltration pond on the surface of a solid waste landfill. The re-circulating liquids are pumped into the pond and are allowed to infiltrate. Optimum operation is for a surface area on the order of 40 m^2 or smaller. In larger modern landfills, this technique is not acceptable due to odor concerns (odors attract birds), aesthetics, safety, rainwater infiltration, operation under cold weather, and others.

6.2 Spray Systems

Liquids can be sprayed onto the landfill in various ways, for example by a tanker truck, a water wagon, a portable tank with fire hose, an irrigation type sprinkler, or others. Tanker trucks or water wagons are typically available at landfill sites for dust control and similar water needs. Pre-wetting of the waste open face tends to improve compaction and is a recommended component of all recirculation programs. The advantage of sprinkler systems is that they distribute the liquid to a relatively large, but controlled area with minimal equipment and labor requirements. The disadvantage is their dependence on weather conditions, because wind conditions can carry odors off site to neighbors, or cause misting toward the landfill staff.

6.3 Horizontal Trenches

Horizontal trenches are constructed by excavating the surface of solid waste, placing a perforated pipe in the trench and backfilling with a permeable material, such as aggregate, fluff, loose waste,

shredded tires, wood chips, crushed glass, or others. Trench systems can be fed by gravity or pressure. Pressure systems seem to operate better because they allow better flow distribution throughout the length of the pipe and benefit from full pipe flow. A typical vertical interval for horizontal trenches is 30-50 feet. Following the installation of the pipe, the trench is covered with additional compacted solid waste. Horizontal trenches have the advantage of uniform moisture distribution within the landfill and are typically the cheapest and easiest to construct method.

6.4 Vertical Wells and Vertical Injection Needles

Vertical wells for leachate recirculation require drilling into the waste mass and installing piping. In some cases, wells are constructed as waste is placed by installing pipe sections at each waste lift. Vertical wells are advantageous for landfills where waste is already in place or where landfill configuration or operation does not permit horizontal trenches. However, the moisture distribution from vertical wells is limited, therefore a large number of wells may be required.

Vertical injection needles are perforated steel pipes that represent a lower cost method, because no drilling is required and the installation is fast. However, vertical needles can accept a limited quantity of liquid.

6.5 Area Infiltration Systems

This technology refers to the installation of layers, typically one to two feet thick, of highly permeable materials (aggregate, tire chips, wood chips, etc.). These layers are meant to provide uniform distribution of re-circulated liquids over the widest possible area. Perforated distribution pipes placed in the permeable zone, vertical wells, gravel columns, or some combination of these components, are used to introduce liquid into the permeable zone. The cost of the permeable material tends to be quite high unless waste streams (e.g. tire chips) can be used for this layer.

6.6 Combined Gas Extraction/Recirculation

It may be possible to combine the piping system used for landfill gas collection with a liquids recirculation system, but such a design would require careful construction, and monitoring procedures to avoid liquids in gas wells that interfere with LFG collection.

6.7 Existing distribution pipe technology

As mentioned in Section 4.2, part of the treated leachate from the leachate collection facility is temporarily sent back to the landfill via a system of distribution pipes. Given the locations of the landfill, the leachate facility and the incinerator (Figure 11), it would be convenient to use this existing technology to send the wash tower effluent from the incinerator to the landfill. Because the leachate facility is downhill of the landfill, pumps are required to transport the liquids into the landfill, therefore it would be cost-effective to use the same pumps and piping systems for a permanent bioreactor operation.

Figure 11: Incinerator to Leachate facility

Currently the transportation system from the leachate facility to the landfill (short arrow in Figure 11) is as follows:

There are two pumps at the leachate facility, which are called Progressive Cavity Pumps (MR 25I5) and have a capacity of 5 m^3/hour, and pressure of 2 bar. The capacity of the pipes is well above the flow of the wash tower effluent (0.25 m^3/hour), therefore it is reasonable to assume the same pipes may be used for the bioreactor operation. Moreover, no mixing with the leachate product stream will be necessary, since once the construction of an evaporator at the leachate facility is complete, the treated leachate will be evaporated instead of recycled. The cross-section of the distribution system within the waste is as follows:

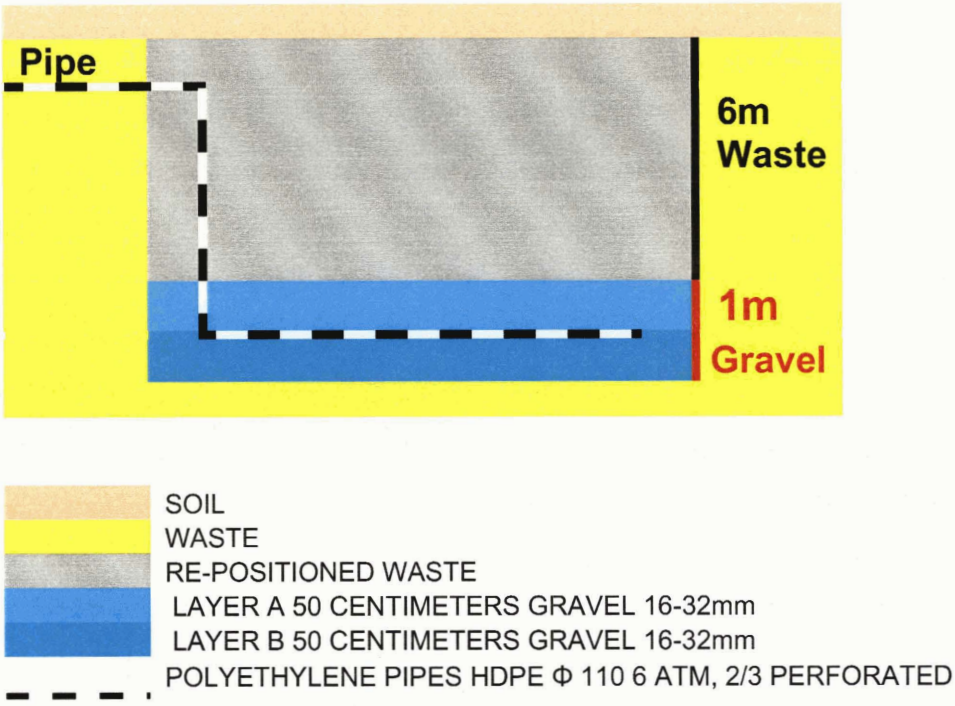

Figure 12: Cross section of Distribution System

The figure above indicates the relative thicknesses of the following layers: soil on top of the landfill, waste that has been removed and the repositioned on top of the gravel layers and the surrounding waste. As shown above, the pipes penetrate the repositioned waste mass and discharge the water into a 1-m thick layer of gravel. The purpose of the pipes is to distribute the water into a wide area within the fill (instead of a point) in order to avoid channelling and ponding within the waste mass. Once the current operation of this distribution system ceases, the wastewater from the hospital waste incinerator could replace the treated leachate and enter the gravel layer in the same manner to enhance permanent bioreaction.

7.0 Conclusion

The results of this study are favorable with respect to operating a small cell of the Ano Liossia landfill of Athens as a bioreactor. We sought to analyze the effects of adding 6 m^3 of wash tower effluent daily to an area of the landfill determined based on this flow rate using the method of mass balance. We addressed the issues of increased leachate production due to addition of wash tower effluent and increased methane production due to the higher rate of biodegradation in bioreactor landfills. Based on the chemical composition of the wash tower effluent, we also analyzed the effects that the wash tower effluent would have on the biodegradation processes in the landfill.

The water mass balance using the Thornthwaite method showed that the increased amount of leachate generated due to bioreaction can be tolerated by the leachate collection system in place. The hydrocarbon-methane mass balance indicated that there is increased generation of methane that can be translated into increased revenue for the landfill gas-energy plant, as well as rapid waste settlement and hence extended landfill lifetime. Finally, our model for predicting hydrogen sulfide generation from the wash tower sulfates suggested that there was not enough wastewater flow for any significant effect on hydrogen sulfide generation to occur. However, the concentration of sulfates in the wastewater would pose a sulfide toxicity threat to the methane-producing bacteria, since it was three times above the threshold representing bacteria inhibition. The non-limiting carbon source however, in combination with the low wastewater flow appear to alleviate the toxic effects of sulfides, thereby allowing the production of methane to proceed as expected.

Our predictions are favorable, however several issues are raised with respect to the applicability of these theoretical results. In terms of the reuse of the recovered landfill space for instance, we do not know how placement of new waste on top of a stabilized waste mass will affect landfill stability. Moreover, we do not know whether the theoretical waste degradation rate values found in the literature for bioreactor landfills can actually be attained in a real situation.

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Appendices

Appendix A: Calculation of landfill area

1 short ton =	907.2 kg				
13000000 gallons of liquid needed for		400000 tons of waste			
According to EPA, a landfill needs to receive		35.82	gallons of water per metric ton of waste, in order to be considered a bioreactor.		
Wastewater flow is	6 m3/d				
	1585 gpd				
So the plant's flow will be sufficient for		44.2	tons/d	64596.25 tons for 4 years	
Density of waste is	0.8 tn/m3				
Volume	55.3 m3/d	for 4 years =>		Volume	80745 m3
Height of cell	57 m				
Area	1416.6 m2				

The above calculation includes the following steps: The wastewater flow is translated into a mass per day according to the EPA requirement (6m3/day => 44.2 tons/day). The landfill operates for 4 years, so the mass per day multiplied by the lifetime of the landfill becomes a mass (64,596 tons). Dividing the mass by the density of the waste, we obtain a volume of waste (80,745 m3). Finally, we divide the volume by the height of the landfill cell and we obtain the area of the cell (1417 m2).

Appendix B: Thornthwaite Water Mass Balance

WATER MASS BALANCE

Conversion factor 1 inch = 25.4 mm
1m = 39.4 in

	January	February	March	April	May	June	July	August	Septemt	October	Novemb	Dec
T air (°C)	8.1	8.7	10.9	15	20.2	25.1	27.4	27	23	17.8	13.4	9.8
I	2.08	2.31	3.25	5.28	8.28	11.32	13.14	12.63	10.08	6.84	4.45	2.77
UPET (inches)	0.02	0.02	0.04	0.06	0.11	0.16	0.19	0.18	0.14	0.09	0.05	0.03
r-PE adjust.	25.5	25.2	30.9	33	36.9	37.2	37.5	35.1	31.2	26.8	25.2	24.9
PET (inches)	0.50	0.60	1.09	2.08	4.07	5.86	7.09	6.36	4.42	2.32	1.29	0.69
P (inches)	2.14	1.88	1.69	1.07	0.83	0.42	0.29	0.19	0.54	1.95	2.26	2.72
Cro	0	0	0	0	0	0	0	0	0	0	0	0
RO	0	0	0	0	0	0	0	0	0	0	0	0
IR, wash tower wastewater input	5.01	5.01	5.01	5.01	5.01	5.01	5.01	5.01	5.01	5.01	5.01	5.01
I, infiltration (inches)	7.14	6.89	6.70	6.08	5.84	5.42	5.30	5.19	5.55	6.96	7.27	7.73
I - PET (inches)	6.64	6.29	5.60	4.00	1.77	-0.43	-1.79	-1.17	1.12	4.64	5.98	7.04
ACC WL (inches)						-0.43	-2.22	-3.39				
Soil moisture capacity (inches)	2											
ST, Storage (inches)	2	2	2	2	2	1.23	0.37	0.18	2	2	2.00	2.00
ΔST, change in storage (inches)	0	0	0	0	0	-0.77	-0.86	-0.19	1.82	0	0	0
AET, actual evapotranspiration (inches)	0.50	0.60	1.09	2.08	4.07	6.19	6.16	5.38	4.42	2.32	1.29	0.69
PERC, percolation (inches)	6.64	6.29	5.60	4.00	1.77	0.00	0.00	0.00	-0.70	4.64	5.98	7.04
Check: P+IR = PERC+AET+ΔST+RC	7.14	6.89	6.70	6.08	5.84	5.42	5.30	5.19	5.55	6.96	7.27	7.73

This is the entire Thornthwaite mass balance for all months. Here is an explanation of the terms in the left column:

T - Air Temperature
I - Heat Index
UPET - Unadjusted potential evapotranspiration
r- PE - Adjustment factor for duration of sunlight
PET – Potential evapotranspiration
P – Precipitation
Cro – Runoff coefficient
RO – Monthly runoff
I – Infiltration
I-PET – Water available for storage
ACC WL – Accumulated water loss
ST – Soil moisture storage
ΔST – Change in soil moisture from last month
AET – Actual evapotranspiration
PERC – Percolation

Appendix C: Results of LandGEM model for Ano Liossia landfill as a bioreactor

Year	Waste Accepted		Waste-In-Place		Methane			
	(Mg/year)	(short tons/year)	(Mg)	(short tons)	(Mg/year)	(m^3/year)	(av ft^3/min)	(Mg/year)
2003	16,149	17,764	0	0	0	0	0	0
2004	16,149	17,764	16,149	17,764	3.401E+02	5.098E+05	3.426E+01	9.333E+02
2005	16,149	17,764	32,298	35,528	5.921E+02	8.875E+05	5.963E+01	1.625E+03
2006	16,149	17,764	48,447	53,292	7.788E+02	1.167E+06	7.843E+01	2.137E+03
2007	16,149	17,764	64,596	71,056	9.171E+02	1.375E+06	9.236E+01	2.516E+03
2008	0	0	80,745	88,820	1.020E+03	1.528E+06	1.027E+02	2.797E+03
2009	0	0	80,745	88,820	7.553E+02	1.132E+06	7.607E+01	2.072E+03
2010	0	0	80,745	88,820	5.595E+02	8.387E+05	5.635E+01	1.535E+03
2011	0	0	80,745	88,820	4.145E+02	6.213E+05	4.175E+01	1.137E+03
2012	0	0	80,745	88,820	3.071E+02	4.603E+05	3.093E+01	8.425E+02
2013	0	0	80,745	88,820	2.275E+02	3.410E+05	2.291E+01	6.242E+02
2014	0	0	80,745	88,820	1.685E+02	2.526E+05	1.697E+01	4.624E+02
2015	0	0	80,745	88,820	1.248E+02	1.871E+05	1.257E+01	3.426E+02
2016	0	0	80,745	88,820	9.249E+01	1.386E+05	9.315E+00	2.538E+02
2017	0	0	80,745	88,820	6.852E+01	1.027E+05	6.901E+00	1.880E+02
2018	0	0	80,745	88,820	5.076E+01	7.608E+04	5.112E+00	1.393E+02
2019	0	0	80,745	88,820	3.760E+01	5.636E+04	3.787E+00	1.032E+02
2020	0	0	80,745	88,820	2.786E+01	4.176E+04	2.806E+00	7.643E+01
2021	0	0	80,745	88,820	2.064E+01	3.093E+04	2.078E+00	5.662E+01
2022	0	0	80,745	88,820	1.529E+01	2.292E+04	1.540E+00	4.195E+01
2023	0	0	80,745	88,820	1.133E+01	1.698E+04	1.141E+00	3.108E+01
2024	0	0	80,745	88,820	8.390E+00	1.258E+04	8.450E-01	2.302E+01
2025	0	0	80,745	88,820	6.216E+00	9.317E+03	6.260E-01	1.705E+01
2026	0	0	80,745	88,820	4.605E+00	6.902E+03	4.638E-01	1.263E+01

This table indicates LandGEM's prediction of the amount of methane (in units of volume and mass) that would be generated if the Ano Lossia landfill were operated as a bioreactor. The column on the left represents the amount of waste that enters the landfill each year of the landfill's operating life, and the column to its right refers to the amount of waste in the landfill at any given time. The maximum amount of methane produced is in the year 2008, and it corresponds to 1,375,000 m^3.

Appendix D: Calculation of the percentage of H_2S in the total landfill gas

Hydrogen sulfide			Total landfill gas		
(Mg/year)	(m^3/year)	(av ft^3/min)	(Mg/year)	(m^3/year)	(av ft^3/min)
0	0	0	0	0	0
9.670E-03	6.822E+00	4.584E-04	2.366E+02	1.895E+05	1.273E+01
1.887E-02	1.331E+01	8.944E-04	4.618E+02	3.698E+05	2.484E+01
2.762E-02	1.948E+01	1.309E-03	6.759E+02	5.412E+05	3.636E+01
3.594E-02	2.536E+01	1.704E-03	8.796E+02	7.043E+05	4.732E+01
4.386E-02	3.094E+01	2.079E-03	1.073E+03	8.595E+05	5.775E+01
4.172E-02	2.943E+01	1.978E-03	1.021E+03	8.175E+05	5.493E+01
3.968E-02	2.800E+01	1.881E-03	9.712E+02	7.777E+05	5.225E+01
3.775E-02	2.663E+01	1.789E-03	9.238E+02	7.397E+05	4.970E+01
3.591E-02	2.533E+01	1.702E-03	8.788E+02	7.037E+05	4.728E+01
3.416E-02	2.410E+01	1.619E-03	8.359E+02	6.694E+05	4.497E+01
3.249E-02	2.292E+01	1.540E-03	7.951E+02	6.367E+05	4.278E+01
3.091E-02	2.180E+01	1.465E-03	7.564E+02	6.057E+05	4.069E+01
2.940E-02	2.074E+01	1.394E-03	7.195E+02	5.761E+05	3.871E+01
2.797E-02	1.973E+01	1.326E-03	6.844E+02	5.480E+05	3.682E+01
2.660E-02	1.877E+01	1.261E-03	6.510E+02	5.213E+05	3.503E+01
2.530E-02	1.785E+01	1.199E-03	6.193E+02	4.959E+05	3.332E+01
2.407E-02	1.698E+01	1.141E-03	5.890E+02	4.717E+05	3.169E+01

These are the results of LandGEM model for a conventional landfill (i.e. no sulfate-rich wastewater is added) and they indicate the amount of hydrogen sulfide and the amount of total landfill gas produced at the landfill. Each line represents a year from 2003 to 2019. The maximum amount of LFG production occurs in year 2007 (boxed values).

To calculate the percentage of H_2S in the total landfill gas by volume, we divided the boxed number on the left table by the boxed number on the right table, to obtain:

$$= \frac{30.94 m^3 / year}{859500 m^3 / year} = 0.0036\%$$

Appendix E: Results of our sulfate reduction model

Input parameters

The model:

$$Q\ H2S = \Sigma\Sigma[k*Lo*(Mi/10)*\exp(-k*t)] + k*Q(SO4)*\exp-k*t)$$

k	0.049	/day
Lo	0.000941038	m3 H2S/ton of waste
Qww	6	m3/day
C (SO4)	975	mg/l
Q (SO4)	2.14	tons SO4/yr
Mi	16149	tons/yr

Year	Total H2S emissions (m^3/year)	H2S Emissions (from waste) (m^3/year)	Emissions (additional) (m^3/year)
2004	0.77	6.540E-01	1.143E-01
2005	1.39	1.277E+00	1.089E-01
2006	1.97	1.870E+00	1.037E-01
2007	2.53	2.434E+00	9.871E-02
2008	3.07	2.972E+00	9.399E-02
2009	2.92	2.830E+00	8.950E-02
2010	2.78	2.694E+00	8.522E-02
2011	2.65	2.566E+00	8.114E-02
2012	2.52	2.443E+00	7.726E-02
2013	2.40	2.326E+00	7.357E-02
2014	2.28	2.215E+00	7.005E-02
2015	2.18	2.109E+00	6.670E-02
2016	2.07	2.008E+00	6.351E-02
2017	1.97	1.912E+00	6.048E-02
2018	1.88	1.821E+00	5.758E-02
2019	1.79	1.734E+00	5.483E-02
2020	1.70	1.651E+00	5.221E-02

The **Total H_2S emissions** column represents the addition of the two columns on its left, which represent the amount of hydrogen sulfide produced from the waste and from the wastewater respectively.