# INFLUENCE OF STAND-ALONE VERTICAL GAS VENTS ON AERATION AND DECOMPOSITION OF ORGANIC WASTE IN MUNICIPAL SOLID WASTE LANDFILLS

タディス, ヨエル ディロン



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# INFLUENCE OF STAND-ALONE VERTICAL GAS VENTS ON AERATION AND DECOMPOSITION OF ORGANIC WASTE IN MUNICIPAL SOLID WASTE LANDFILLS

By

TADIS DILLON

A Thesis Submitted

In Partial Fulfillment of the Requirements

For the Degree of

**Doctor of Engineering**

Examination Committee

Prof. Takayuki Shimaoka (Chairman)

Prof. Takahiro Kuba

Prof.Hemanta Hazarika

九州大学

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DEPARTMENT OF URBAN AND ENVIRONMENTAL ENGINEERING
GRADUATE SCHOOL OF ENGINEERING
**KYUSHU UNIVERSITY**
Fukuoka, Japan

CERTIFICATE

The undersigned hereby certify that they have read and recommended to the Graduate School of Engineering for the acceptance of this thesis entitled, ***"Influence of stand-alone vertical gas vents on aeration and decomposition of organic waste in municipal solid waste landfills"*** by **Tadis Dillon** in partial fulfillment of the requirements for the degree of **Doctor of Engineering**.

Dated: December, 2023

Thesis Supervisor: ____________________
Prof. Takayuki SHIMAOKA, Dr. Eng.

Examination Committee: ____________________
Prof. Takayuki SHIMAOKA, Dr. Eng.

____________________
Prof. Takahiro KUBA, Dr. Eng.

____________________
Prof.Hemanta Hazarika, Dr. Eng.

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# ABSTRACT

With rapid development and industrialization waste generation by society has also increased. It is estimated that currently 2 billion tons of waste is generated per annum. Based on its source, the composition of waste differs. A typical solid waste sample contains, in varying proportions, items such as wood, food waste, garden waste, paper, packaging materials, clothing, construction waste, glass, plastic and metals. In developing countries putrescible wastes, such as food and garden waste, make up most of the waste of these countries. Developing countries generally have a higher fraction of organic waste which may account for over 50% of disposed municipal solid waste. Of the waste generated annually it is estimated that only about 33% is disposed of in a sustainable manner. This has resulted in many uncontrolled dumping sites being utilized in developing countries. For waste, which is disposed of properly, landfills are the most widely used form of final waste disposal. However, in many developing countries these landfills, due to the high organic content of the waste, pose a risk to society from untreated leachate and greenhouse gases. Because of the dominance of landfills as final disposal is beneficial to study techniques which can be employed to reduce the impact from landfills with organic wastes. Techniques have been developed such as aerobic and semi-aerobic landfilling which aim to introduce oxygen into the landfill to accelerate the decomposition process. While effective these methods suffer some setbacks which will make their widespread implementation in developing countries limited. Ventilation shafts placed into waste have been implemented as a means of landfill gas management for the release of gases from the landfill to the atmosphere and are effective at this task as they are a low cost, low maintenance and passive. However, their use has been primarily focused on their application to gas release and not on their potential use for oxygen introduction into landfill masses.

Chapter 1 introduces the overview of the global status of municipal solid waste (MSW) and landfills including definitions, generation, composition and collection. Treatment methodologies and disposal techniques were then explored. Finally, the objectives and structure of the study were presented.

Chapter 2 investigated how strong organic content can impact on the oxygen distribution and consumption rates in a waste where respiration by microfauna becomes the dominant oxygen sink. Further it investigated the effect of varying organic content on oxygen distribution and consumption dynamics. Immature waste bodies were observed to produce highly variable oxygen consumption and distribution due to unpredictable microfauna proliferation. Increasing organic content elevates oxygen consumption but the increased consumption is not proportional to the increase in organic availability. These factors suggest that the oxygen consumption rate in immature, highly organic, waste is not simply predicted based solely on time. It may be acceptable to assign a generalized, fixed rate of oxygen consumption for highly organic landfills as the rate may vary both temporally and spatially at a single site, yet the maximal rate may not differ significantly between sites of different organic loading.

Chapter 3 attempted to recreate landfill conditions to analyze the result of implementation of passive ventilation systems in landfills. Two-dimensional(2D) lysimeters were established to investigate the effectiveness of stand-alone, vertical ventilation pipes inserted into waste masses. Two different

configurations of ventilation were tested with the third lysimeter acting as an unventilated control. Lysimeters were left uninsulated and observed over the course of six months with regular collection of gas and leachate samples. Lysimeters were then simulated for Oxygen ($O_2$) and Nitrous oxide ($N_2O$) to analyze the denitrification contributions of each. The experiment revealed that a single ventilation pipe can increase the mean oxygen level of a 1.7m*1.0m area by up to 13.5%. It also identified that while increasing the density of ventilation pipes led to increased $O_2$ levels, this increase was not significant at the 0.05 probability level. A single vent averaged 13.7% $O_2$ while inclusion of an additional vent in the same area only increased the average to 14.6%, a 6.7% increase. Simulation helped to verify that lower ventilation pipe placement density may be more efficient as in addition to the effect on oxygenation, denitrification efficiency may increase.

Chapter 4, a model was developed to simulate the oxygen transport and behavior observed from the two-dimensional lysimeter experiments. The computer simulation package, COMSOL Multiphysics, was successfully used to output a simulation which could satisfactorily fit the experimental data from the lysimeters. The simulation results from the calibrated model were then utilized to undertake a larger scale simulation of a theoretical landfill cell with similar conditions. This model was able to show that, the introduction of passive ventilation shafts can improve an anaerobic landfill. This was based on prolonged increased levels of oxygen being available which had the ability to induce the aerobic degradation for a longer period which in turn generated more heat that increased microbial activity which also accelerated anaerobic processes as well. This could have benefits in reducing the aftercare period needed for closed landfills and reducing the duration of greenhouse gas emissions from methane.

Chapter 5, conclusions based on the research and the recommendations for future studies on optimization of the passive aeration were proposed.

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# CHAPTER 1

# INTRODUCTION

# CHAPTER 1

## 1. Introduction

## 1.1 Global waste management

### 1.1.1 MSW definition

An unavoidable externality created from the activities of man is the generation of by-products. In a perfect system these by-products would become the inputs to other processes, creating a closed loop. However, with rapid development and industrialization it is becoming more difficult to close the material cycle (Letcher & Vallero, 2019). As such, now these byproducts have been given the generalized term waste which can be defined as unwanted or unusable materials which are derived as a from personal or economic activities. Waste can be generated in different forms including liquid, gaseous, semi-solid and solids.

Within the solids category, municipal solid wastes (MSW) are included. These are solid wastes which are generated within a clearly identifiable area or municipality and may include waste generated by commercial, industrial, household, or individual activity. However, it excludes hazardous waste(Rao et al., 2016).

### 1.1.2 MSW composition

Based on its source, the constitution of MSW varies. A typical MSW sample contains, in varying proportions, items such as wood, food waste, garden waste, paper, packaging materials, clothing, construction waste, glass, plastic and metals. These waste types can be simply grouped into two main categories: Putrescible and non-putrescible waste. Putrescible wastes can be defined as wastes which can be easily biodegraded by microorganisms as a carbon energy source for their respiration. This category would incorporate wastes such as food, kitchen and green waste. Waste which do not fall into the putrescible waste category can be grouped together as non-putrescible waste. It includes metals, glass, earthen materials (stone, sand, etc.) and plastics. Plastics are included due to their low biodegradability in nature even though they have very high concentrations of organic carbon.

The individual contribution of each type of waste in any given sample of MSW varies both on a global and on a regional scale. This is due to differences in climate, resources, technological development, culture and consumer patterns(Cheremisinoff, 2002). Developing countries generally have a higher fraction of organic waste which may account for over 50% of disposed MSW. This high level of waste production can be attributed to the lower levels of policy and technological advancements. In developed nations advancements have allowed for the reduction of the quantity of organic waste entering the MSW stream. As such, in developed nations, the organic waste component of MSW is reduced to approximately 32%. Figure 1-1 presents a summary of the global waste composition.

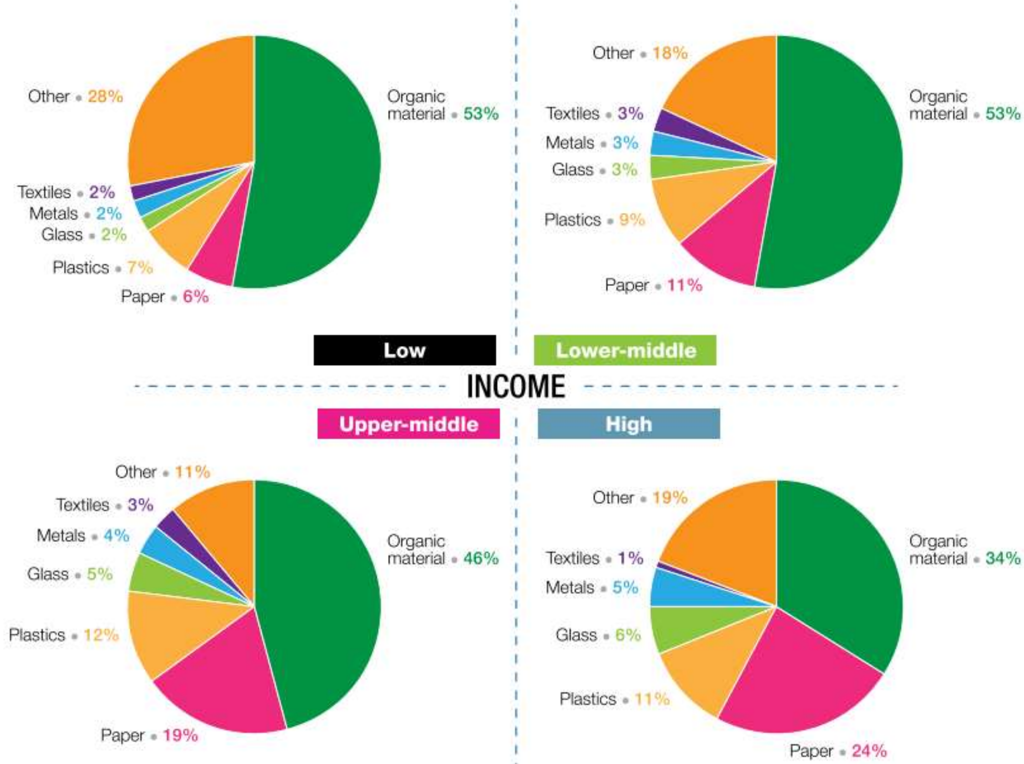


Figure 1-1. Global waste composition sorted by economic status (Wilson et al., 2016)

### 1.1.3 MSW generation

Data presented by the World bank states that globally MSW is generated, on average, at a rate of 0.74kg/day/capita. This amounts to an annual total of approximately 2 billion tonnes. Although technological advancements are improving with time waste generation is not expected to be reduced greatly. This is because by 2050 global waste generation is projected to rise to 3.4 billion. Geographically this waste generation is not evenly distributed with high income/ developed nations, contributing 34% of the global total despite only containing 16% of the global population. Developing regions, even though containing most of the world's population, currently contribute lower than the 0.74 kg/day/capita daily average. However, projections estimate that the waste of some low income, developing countries, is expected to triple by 2050. These projections are presented in Figure 1-2.

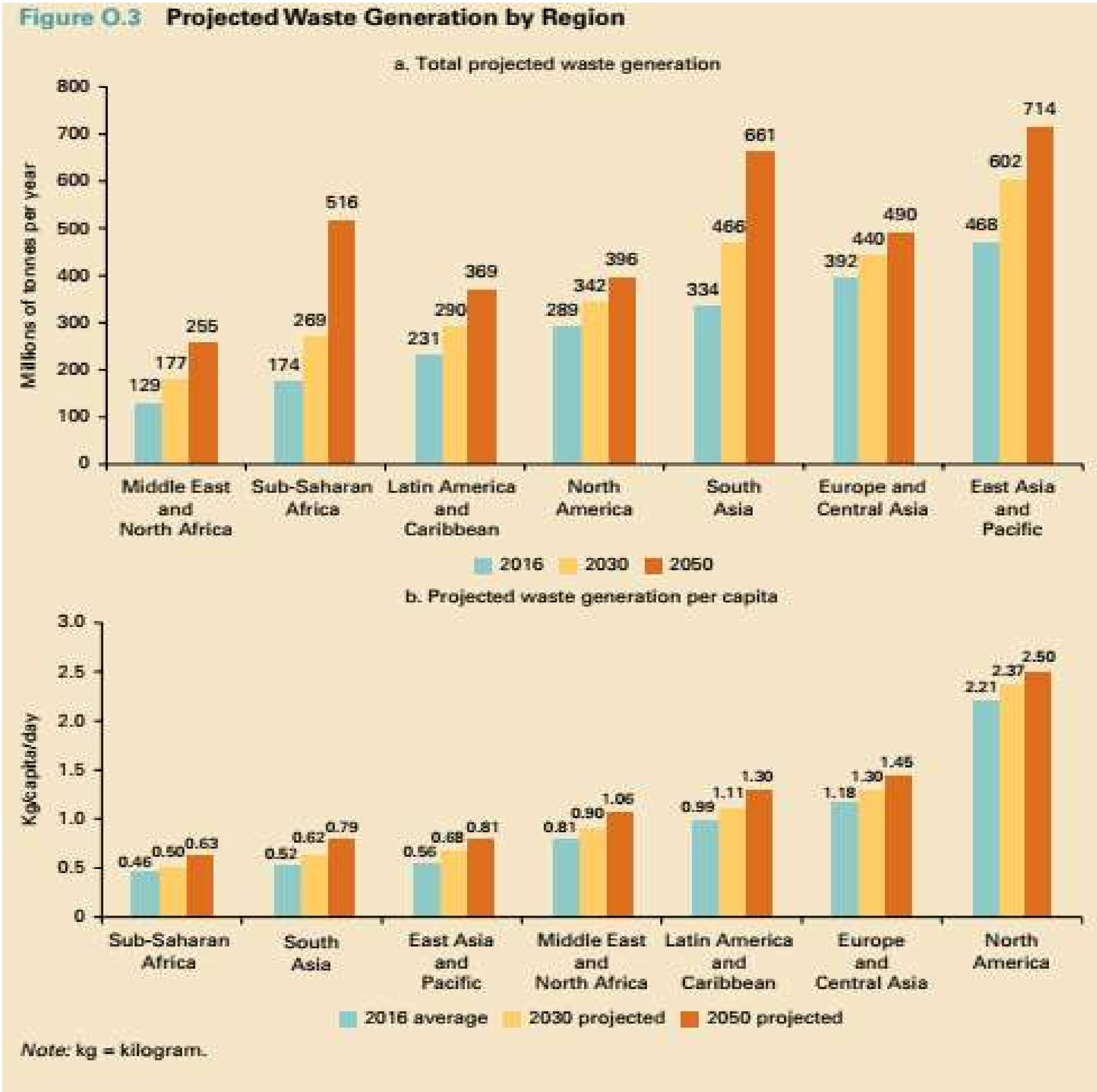


Figure 1-2. Present and forecasted global waste generation statistics. (Kaza et al., 2018)

### 1.1.4 MSW management

Globally, waste management has been guided by a general framework known as the waste management hierarchy. This hierarchy recommends that higher level actions, which reduce the generation of MSW, should be prioritized and disposal of waste should only be the means of last resort after all other available measures have been pursued. The waste management hierarchy can be seen in Figure 1-3. In recently times, there has been a drive led by the United Nations to move from the concept of waste management to one of resource management. With this concept waste is not considered as unwanted but as a resource which can be utilized in other process. This way of thinking is in line with the waste management hierarchy as final waste disposal of a becomes a last resort after all valuable uses have been pursued and no further economically viable extractions can be made. This concept has been given the term circular economy (Wilson et al., 2016).With the circular economy it can be expected that waste products from one process or industry will become a feedstock for another process. Alternatively with the circular economy concept there will be an incentive to alter existing processes which produce unusable byproducts into ones which produce useful byproducts. Figure 1-4 outlines an example of the circular economy concept. Regardless of how waste management is addressed by a society, be it by utilizing the landfill hierarchy or implementing a higher-level circular economy the outcome remains the same; some form of waste will be generated requiring final disposal.


Figure 1-3. Waste Management Hierarchy (Wilson et al., 2016)

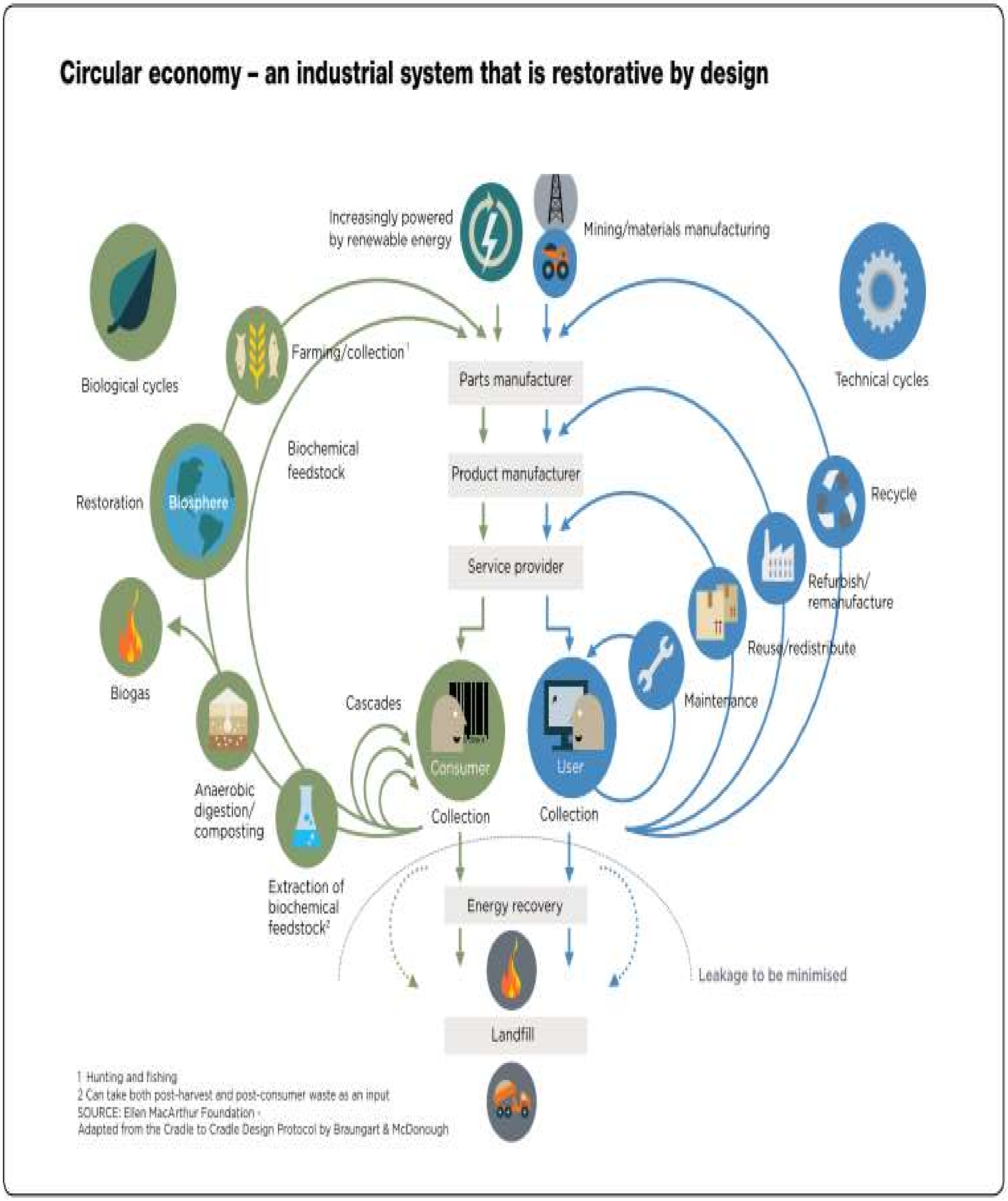


Figure 1-4. Circular Economy Concept. (MacArthur, 2013)

### 1.1.5 MSW treatment

Of the 2 billion tons of MSW generated annually it is estimated that about 33% is disposed of in a sustainable manner. This is attributed to deficiencies in waste collection systems. Waste collection, in most regions, is coordinated by local governments or special purpose entities for waste management. For these coordinating bodies waste collection occupies a significant share of their budgets (Letcher & Vallero, 2019). As such collection efficiency becomes correlated to economic status as higher income countries can allocate more resources. In many low income , developing countries waste collection only covers around 40% of the population and increases to 51% and 82% in middle income and developed countries respectively (Breukelman et al., 2019). Urban populations generally have better collection rates as collection efforts become less efficient in rural areas where it is less economical to assign limited resources over a large geographic area.

Open dumping of waste accounts for about 33% of global waste disposal, 19% is recovered through recycling and composting, and 11% is incinerated for final disposal. The final 37% of waste is disposed of in controlled landfills. Figure 1-5 shows the breakdown of controlled landfilling statistics based on economic status. With only 35% of their waste going to controlled landfill sites developing countries are the main contributor to the open dumping problem. As most of the waste is not entering controlled sites most of the waste disposed by developing countries is treated in an unsustainable manner.

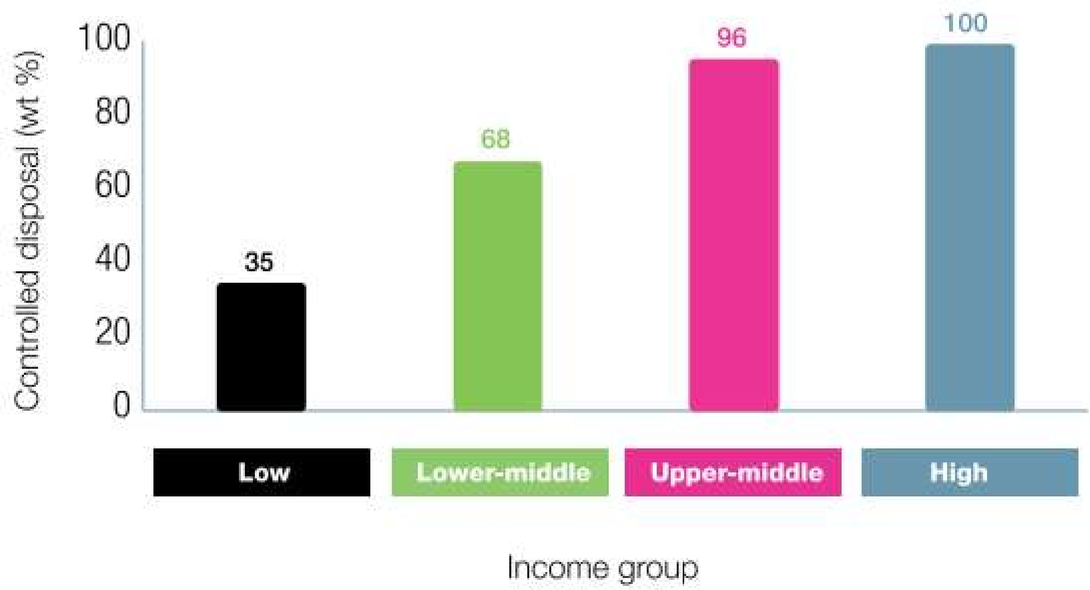


Figure 1-5. Global controlled disposal statistics. (Wilson et al., 2016)

### 1.2 Sustainable treatment methodologies

There are two feasible possibilities for the final disposal of MSW: Ocean or Land Disposal. With few exceptions most countries do not permit the legal disposal of MSW into the oceans (Worrell & Vesilind, 2012) As such the only remaining option is disposal to land. Because of the limited options landfills have become the most widely used form of final waste disposal. Although there are a range of landfilling methods there are few methods which are accepted as controlled, environmentally sound techniques. The controlled techniques have been categorized based on the chemical degradation pathway which predominates. Figure 1-6 outlines the main degradation pathways in landfills.

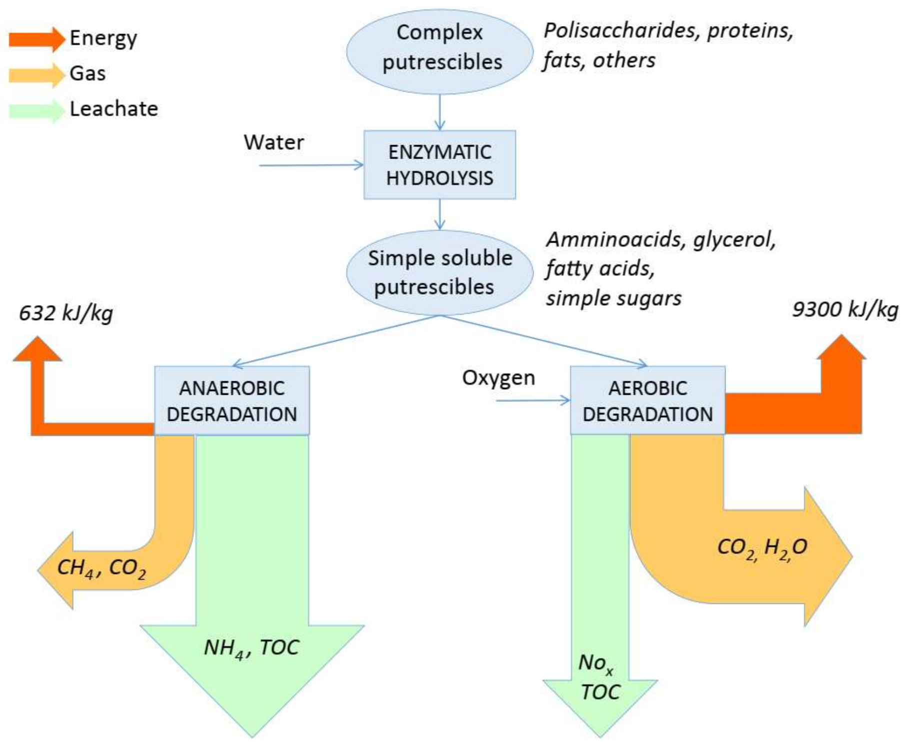


Figure 1-6. Landfill degradation processes. (Cossu et al., 2018)

### 1.2.1 Anaerobic Method

Anaerobic landfills were the first type of controlled landfills which became utilized widely. These landfills were the natural evolution from open dump sites which have been utilized before the environmental risk of uncontrolled disposal were known. They began with careful site selection to ensure the selected area for the landfill had natural impermeable clay barriers and would not impact on ground water sources. Over time they have they have evolved to include synthetic liners in additional to natural site selection. Once the sites were selected leachate collection systems were installed to ensure leachate could be collected. This prevented excessive leachate heads from forming in the waste as well as allowed this leachate to be treated. Further protections came from implementing landfill operation techniques of spreading and compaction of waste followed by utilizing soil to cover each layer. This was implemented to reduce the nuisance impact of waste due to odors and pests. These actions together result in landfills which become isolated from the atmosphere which results in waste undergoing anaerobic respiration with the main by-product being methane gas. The anaerobic pathway can be seen in Equation 1-1 where:

$$C_6H_{12}O_6 \text{-->} CH_4 + CO_2 + \text{Biomass} + \text{Heat} \qquad \text{(Eq1-1)}$$

In recent times anaerobic landfills have been repurposed into anerobic bioreactors where the landfill operations are altered to maximize methane generation and collection. This methane is then utilized to generated either thermal or electrical energy which can be reused onsite or sold to a grid.

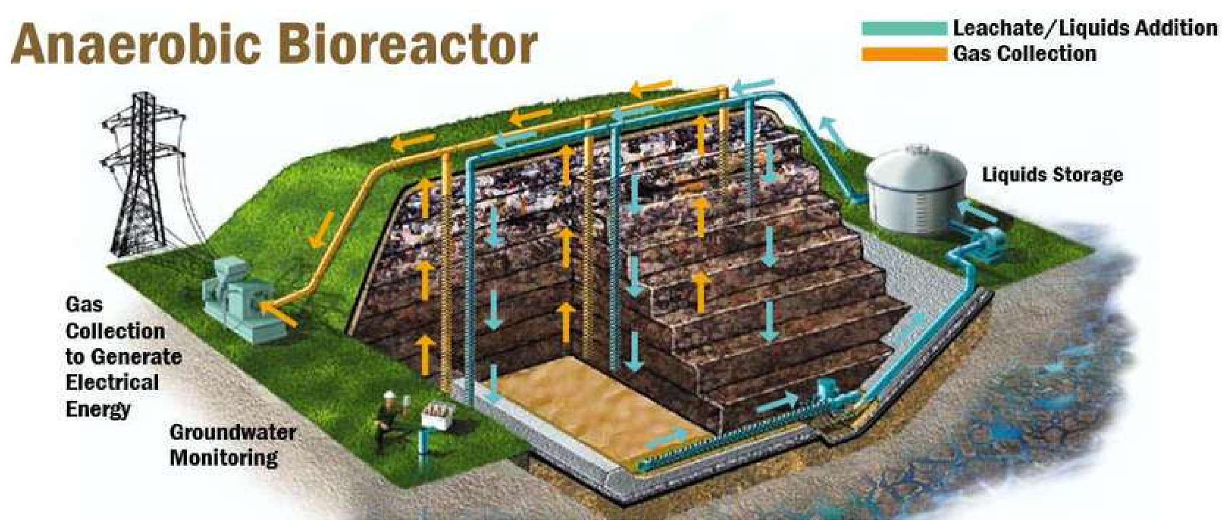


Figure 1-6. Modern Anerobic Landfill. (Waste Management, 2014)

### 1.2.2 Aerobic Method

The aerobic landfill concept came into focus after gas extraction from anerobic landfills became popularized. At some point gas extraction from anerobic landfills becomes unfeasible yet these landfills are still in need of after care as they are not fully stabilized. Landfill aeration was then conceived as tool for the rapid stabilization of these anaerobic sites. (Ritzkowski & Stegmann, 2012). With aerobic landfills air is actively moved into the landfill mass via pipes and pumps. With an abundance of air available the predominant mode of respiration of bacteria shifts from anaerobic degradation to aerobic degradation. The aerobic pathway can be seen in Equation 1-2 where:

$$C_6H_{12}O_6 + O_2 \dashrightarrow CO_2 + H_2O + \text{Biomass} + \text{Heat} \qquad \text{(Eq1-2)}$$

In positive aeration air is pumped into the waste directly while in negative aeration is induced by the active removal of landfill gas which enables the entry of air from other pipes to replace the removed gases.

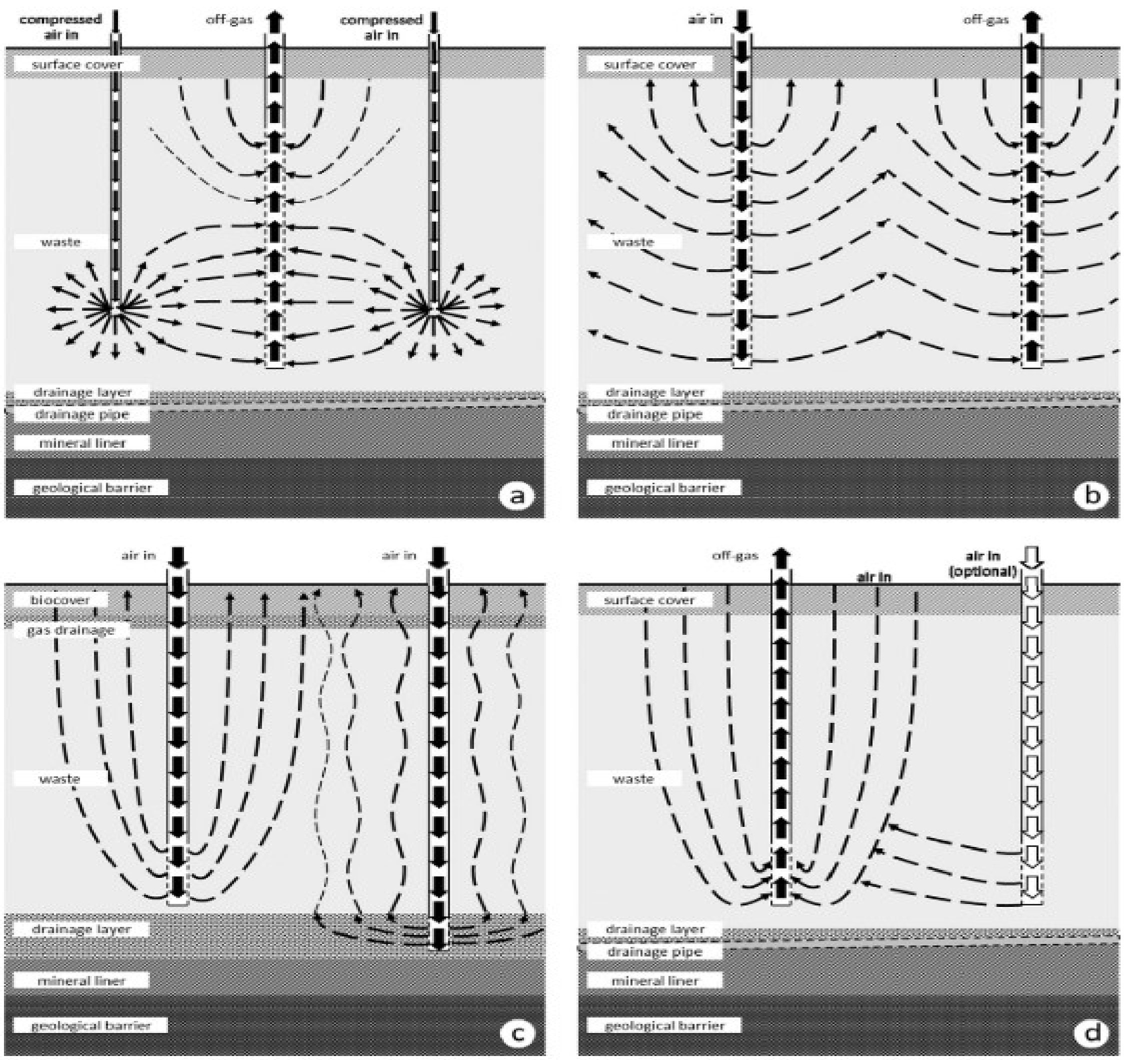


Figure 1-7. Schematic outlines of different aeration concepts. (M. Ritzkowski & Stegmann, 2012)

### 1.2.3 Semi - Aerobic Method

In 1975 a new landfill type was created in Japan. This landfill type originated from anaerobic landfills with the key difference being the leachate collection pipe. In semi-aerobic type landfills air is supplied spontaneously via the leachate collection pipe that has bigger diameter than those utilized in anaerobic landfills. As such the leachate collection pipes in these landfill types fulfill a dual role, the discharge of leachate together with supplying of air. The energy source for the movement of the air from the leachate pipe to the landfill body is the via convective forces generated from air rising due to being heated by the thermal energy generated from waste decomposition.(Shimaoka, T., Matsufuji, Y., & Hanashima, 2000)

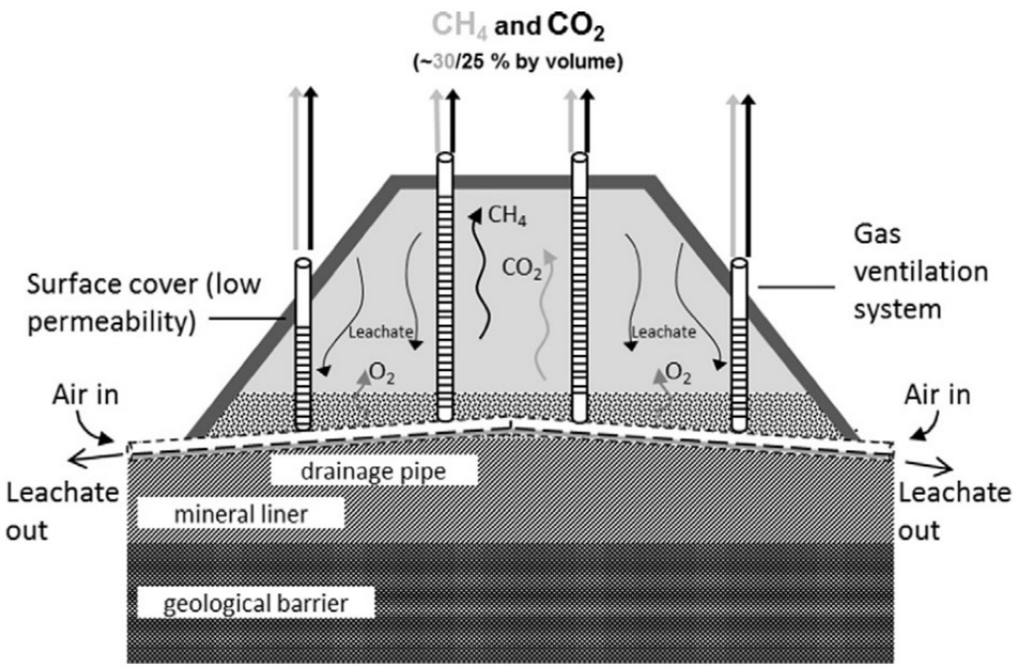


Figure 1-8. Semi Aerobic Landfill. (Ritzkowski & Stegmann, 2012)

### 1.3 Proposed Action

#### 1.3.1 Passive aeration

Passive aeration, with respect to landfills, involves the natural permeation of ambient air into a landfill mass without the expenditure of electrical energy. The concept of passive aeration was born from the basic approach of air venting where gas is exchanged between the atmosphere and landfill body via the surface or through open gas wells. Unlike traditional gas venting which is optimized of removal of gas from the landfill body to the surface, passive aeration is designed for the transfer of air from the ambient environment to within the landfill. For this reason, the gas wells utilized are configured to only be perforated at the end of the pipe which comes into deepest contact with the waste. The reduces short cutting where the air will take the path of least resistance to diffuse from high concentration to low concentration. With only limited perforations available at the base of the pipes, shortcutting is eliminated, and air distribution is optimized.

#### 1.3.2 Implementation

Passive aeration by a proprietary system known as DEPO+® has been reported to have been implemented at landfills in Germany. However, post implementation no data has been discovered about the effectiveness of this system (Ritzkowski & Stegmann, 2012). In Japan in situ landfill studies have also been undertaken where data for gas composition, gas flow and temperature were collected and analyzed (Kim et al., 2010).

### 1.4 Research objectives and outline of dissertation

Globally, traditional landfills are, by far, the most popular form of final waste disposal. A high proportion of food and green wastes is typically present in MSW, especially in developing countries. Traditional anerobic landfills are unable to adequately treat with organic waste which results in leachate with high biological oxygen demand (BOD) and chemical oxygen demand (COD) levels. In countries where there is inadequate or absent leachate treatment systems these leachates can pose a threat to the natural waterways when left untreated.

With the advent of landfill methods such as aerobic and semi-aerobic landfills it has become possible to reduce the pollution load of produced leachates and enable landfills to act as treatment bodies and not just as the location of final storage.

The majority of MSW generated by developing countries is putrescible. This waste can be ideally isolated from entering the waste stream, at source, and treated separately without entering landfills. Treatment options include technologies such as composting and waste to energy technologies including bioreactors and incineration. For these technologies to be cost effective there needs to be an effective, high-quality feedstock available. Waste separation at source would be required in addition to a robust collection system. With waste collection currently pre-occupying a large portion of the budget of local authorities the introduction of additional collection requirements may seem burdensome. Currently the collection systems of all countries are not 100% efficient and in developing countries they are highly inefficient. Other interventions, at other levels of the waste hierarchy previously outlined, are thus needed in conjunction to higher level interventions. Looking at the final level, treatment and disposal, the semi-aerobic landfill method proved to be a highly adoptable technology for developing countries due to its low cost, low maintenance and ease of implementation. However, it requires the establishment of a new landfill cell to be implemented. Currently 33% or more of current MSW is not being disposed of in a sustainable manner. These wastes are thus being disposed of into open dumpsites or uncontrolled landfills. If such sites continue to be used, as is, it can be assumed that the infrastructure needed to implement the semi-aerobic method is not available at these sites. Due to the economic status of the regions where these unsustainable sites are located it can also be assumed funding is not available or is not prioritized for the retrofitting of these sites to become proper sanitary landfills. However, these sites should not be overlooked for rehabilitation. A simple landfill rehabilitation technique, which has been implemented prior, is the use of vertical gas venting pipes inserted from the final cover of landfills. This technique was initially implemented as a means of reducing the buildup of methane gas which is created in the latter stages of the life cycle of an anaerobic landfill. Later it was shown that the implementation of these standalone vertical vents also allowed for the infiltration air into the landfill body as well.

The most prominent research identified on the effectiveness of stand-alone gas vents was the work undertaken by (Kim et al., 2010). This work was considered significant due to the in-situ implementation of stand-alone vents at a closed landfill site with regular surveys undertaken over a 3-year period. During the survey's, data for gas composition, gas flow, and temperature were collected. It was noted that these vents were implemented in the closing stage of the landfill which had been operating for twenty years. Also, the waste composition was composed of only 34.4% MSW with most of the waste deposited consisting of a combination of construction waste and industrial waste (32% and 13% respectively). Over 20 years it may be reasonably expected that putrescible waste would be metabolized over a long period by anaerobic processes. Further, long-term leaching would remove significant organic matter as well. In waste which has been decomposed over an extended period, the rate of oxygen diffusion into the landfill may exceed oxygen depletion from microbial activity due to the absence of sufficient substate. This has led to an aerobic stage in landfills reemerging as over time an anaerobic landfill is thought to eventually become aerobic.(Bozkurt et al., 2000; Kjeldsen et al., 2002; Mønster et al., 2019). As such given the age and the waste composition of the landfill utilized by Kim et al. the potential air intrusion which may be possible given sufficient substrate to promote aerobic degradation still warrants investigation. Little has been studied on the effect the introduction of these stand-alone vents would produce on waste which were not anaerobically decomposed and/or, still possessing abundant organic carbon concentrations.

This thesis serves to evaluate the quantifiable benefits of the introduction of stand-alone vertical ventilation shafts into an unsanitary landfill such as those currently active in developing countries.

Chapter 2 firstly evaluates the effect of organic concentration on the dynamics of passive oxygen diffusion through wastes via a traditional column lysimeter study.

Chapter 3 evaluates the effect of the presence of vertical ventilation pipes have in a waste body via a 2D lysimeter study.

Chapter 4 utilizes a numerical simulation to replicate the lysimeter study and generate a landfill model appropriate for the evaluation of gas transport.

Chapter 5 presents the recommendation for the implementation of stand-alone vertical ventilation systems based on the outcome of simulations performed utilizing the landfill model developed.

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# CHAPTER 2

# OXYGEN CONSUMPTION IN LANFILLED ORGANIC SOLID WASTES

# CHAPTER 2

## 2. Oxygen consumption in landfilled organic solid wastes

2.1 INTRODUCTION

2.2 METHODOLOGY

2.3 RESULTS AND DISCUSSION

2.4 CONCLUSIONS

## 2.1 Introduction

According to the World Bank, in developing countries in Latin America and the Caribbean (LAC), as much as 95% waste generated in these countries is treated by some form of in-situ land disposal on a spectrum ranging from planned sanitary landfills to ad hoc open dump sites(Kaza et al., 2018). Regardless of the location or condition of these disposal sites, they share an important characteristic; as much as 70% of the waste they contain can be considered as some form of easily degradable organic matter (kitchen waste, paper products, wood). With the high organic composition of the wastes of these landfills the introduction of oxygen into these waste bodies is expected to greatly accelerate the stabilization of these sites. While there has been research into understanding the mechanisms affecting the influx of oxygen into soils(Cook, 1995; Cook & Knight, 2003; Neira et al., 2015) the research has been primarily focused the role of flora on oxygen demand. Landfills in LAC and other developing regions pose a different challenge to the soils in these studies due to the expected difference in organic concentration. Studies on understanding and modelling the transport and consumption of oxygen specifically within waste bodies has been previously undertaken(Kallel et al., 2003, 2006). These studies demonstrated the link between oxygen consumption and oxygen diffusion into solid waste. However, the conditions replicated mirrored that of the present municipal solid waste of a developed nation composed predominantly of incineration residues. This waste composition does not reflect the waste composition of the developing world. To understand the oxygen dynamics of organic waste additional work is required. The aim of the current research is to focus on waste of with putrescible organic content. Column experiments were performed to analyze the oxygen profile of putrescible organic waste and the associated oxygen consumption rates.

### 2.2 Materials and Methodology

#### 2.2.1 Samples

Proxy materials were utilized to represent the target waste stream. Easily degradable organic matter in the form of compost (CP) generated from food waste was selected as the organic fraction as it was deemed a close approximate to the kitchen waste. The inorganic and not easily degraded fraction of waste was replicated utilizing waste incineration bottom ash (BA) sieved for the fraction under 10mm then air dried before use.

#### 2.2.2 Materials

Figure 1 shows the layout of the materials utilized for the study. The columns used were composed of acrylic material of dimensions 0.1m (Diameter)* 2m (Height) consisting of two layers; a. Drainage layer, b. waste Layer. The drainage layer consisted of glass beads to facilitate separation and removal of leachate from the waste mass. The waste layer spanned a depth of 187cm and was constructed utilizing CP and BA. BA and CP were subjected to analysis including particle size distribution, moisture content (ω [%]), ignition loss [LOI (mg [g-dry solid (DS)], oxygen consumption and leachability characteristics. The methodologies utilized for these analyses were referenced (Kallel et al., 2003). A total of three (3) columns were constructed with the proportions of 0%CP:100%BA in column 1; 10%CP:90%BA in column 2 and 20%CP:80% BA in column 3. The mixture was placed into the columns in 10cm layers at a time to ensure uniform compaction to an average density of approximately 1100 kg/m3. The columns were purged of oxygen by opening all ports and applying a positive pressure of pure nitrogen gas ($N_2$) to displace oxygen. The addition of nitrogen was terminated and all ports and openings, including the leachate collection port, closed and the baseline oxygen levels determined. Once baseline levels were obtained, the top of each column was opened to allow unrestricted gas exchange.

#### 2.2.3 Sampling

Closed gas sampling ports were established at every 0.1m height intervals for the first meter of the column and subsequently at 0.2m intervals for the remaining meter for a total of fourteen (14) points. Each sampling port was protected by a ball valve. The mouth of each valve was sealed by a plastic sleeve and capped with a rubber cork to isolate the sampling ports from the outside air, when the samples were being removed. Samples were removed utilizing a needle affixed to a 6ml syringe via a 3-way stopcock. Before samples were taken the residual air within each sample port was first withdrawn utilizing the syringe to create a vacuum within the sample port. The needle was then reinserted into the vacuum before the valve was opened. The air within the cavity was then mixed by pumping the syringe repeatedly approximately 10 times. A sample, not exceeding 2-3ml was extracted and kept for analysis utilizing a SHIMADZU GC-2014. Gas samples were obtained from the column initially every two days for the first week, after which they were taken at 5–7-day intervals.

Leachate was collected via collection bottles attached to the column via hoses. When leachate was not

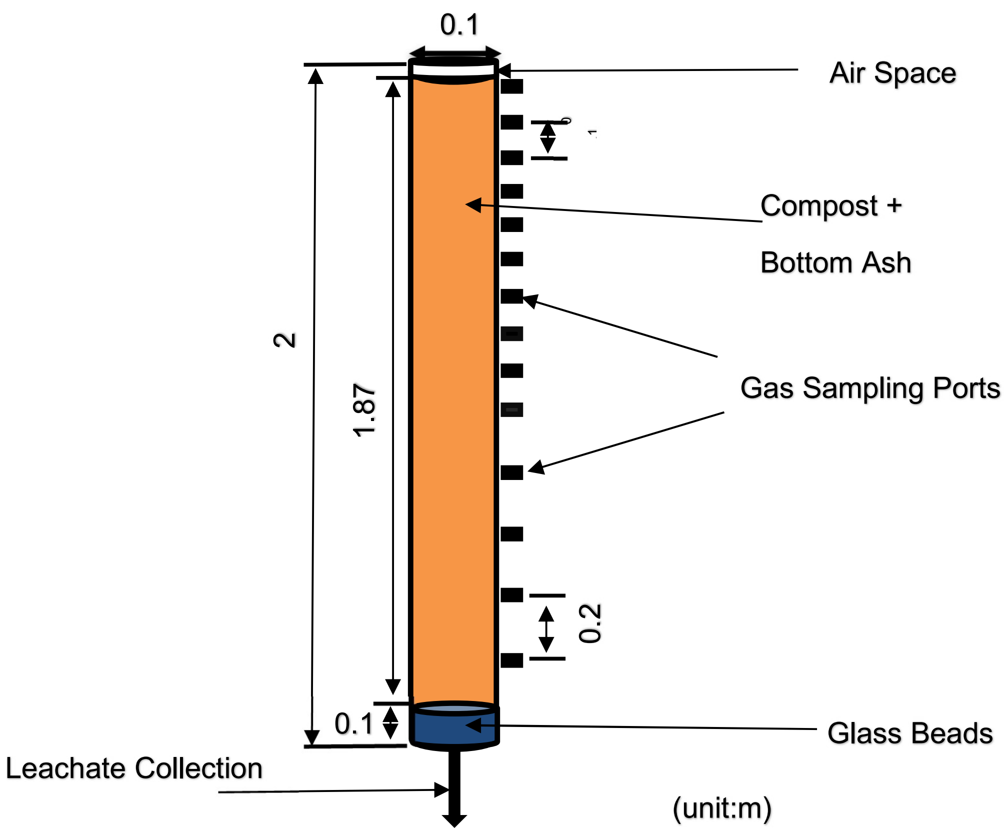


Figure 2-1. Column Experiment Diagram

Figure 2-2. Column Experiment Setup.

being collected, the hoses were sealed to prevent gas entering the column. A leachate head was maintained within the collection system to prevent entry of air when leachate was collected. Leachate was removed on an as needed basis as the drainage layer approached capacity. Collected leachate was recirculated back into the columns.

Moisture was supplied to the columns initially in the form of distilled water at a rate of 40ml per day to simulate an average daily rainfall of typical of tropical climates.

## 2.3 Results and Discussion

### 2.3.1 Waste Characteristics

Table 2-I. Loss on Ignition Test

| Sample | Moisture Content (%) | Loss on Ignition (%) |
|---|---|---|
| BA | 11.88 | 0.78 |
| CP | 11.37 | 60 |

Table 2-II. Leaching Test Analysis

| COLUMN | TOC (mg/L) | IC (mg/L) | TC (mg/L) | TN (mg/L) |
|---|---|---|---|---|
| 0%CP: 100%BA | 19.70 | N.D | 19.7 | 3.20 |
| 10%CP: 90%BA | 383.3 | 0.6 | 383.9 | 49.9 |
| 20%CP: 80%BA | 1411 | 7.0 | 1418 | 178 |

(N.D – Not Detectable)

Tables 2-I and 2-II show results of analysis undertaken on the waste materials utilized. The CP utilized as the organic fraction contained around 60% volatile organics based on loss on ignition tests (2-I). BA utilized contained less than 1% volatile organic matter. Leaching tests (2-II) further demonstrate the low levels of readily available carbon available in BA as compared the elevated levels in CP.

### 2.3.2 Oxygen Distribution

Figures 2-3 to 2-5 present the oxygen distribution profiles derived from the experiment. The first graphic (a) maps the oxygen concentration measured at each sampling point (vertical axis) vs time (horizontal axis). The oxygen concentration at the intervals which were not measured were interpolated based on the measured data. Blue represents areas of low oxygen concentration with red representing atmospheric concentration. Graphic (b) represents the change in oxygen concentration (vertical axis) with time (horizontal axis) at selected sampling ports. Graphic (c) highlights the vertical oxygen distribution within the column on selected days. After air was introduced, it rapidly diffused to the lower levels driven by the large diffusion gradient between the atmosphere and the waste mass created by the oxygen purging at the start of the experiment. In pure BA, oxygen penetrates slowly and continuously driven primarily by the diffusion gradient with resistance to the diffusive movement defined by the diffusion coefficient of oxygen in BA. Within 100 days the diffusion gradient became minimal with no significant difference in oxygen levels between the column surface and the base. In the presence of organic matter the effect on the oxygen profile is readily observed. After initial replenishment of the oxygen concentration to the 12-14% range the subsequent 30 days saw the oxygen levels in both columns begin to decrease. The reduction in oxygen in the columns with high organic matter as opposed to no reduction in bottom ash was thus attributed to oxygen consumption from bacterial respiration utilizing the available substrates. The oxygen distribution profile of organic waste however produced an irregular structure where the oxygen consumption appeared adhoc and oxygen concentration was not linearly distributed as observed in bottom ash. This was attributed to the tendancy of microrganisms in soils ,even though being present at $10^6$ ->$10^{12}$ individual cells per gram of soil, to mainly associate into colonies which in turn develop into respiration hotspots (Schlüter et al., 2019). These hotspots are defined as small pockets within the soil which display accelerated microbial process as compared to the surrounding areas(Kuzyakov & Blagodatskaya, 2015). These hotspots become active during what are termed hot moments; triggers which spur microorganisms moment. Bearing similar characteristics to soils, the waste mass was believed to be subject to a similar process. One dominant factor which is proposed as the catalyst for the promulugation of the hot moments observed in the simulated waste mass was temperature. Due to the columns being composed of acrylic material there was no thermal insulation and as such the column temperatures were influenced by the external conditions. When the atmopheric temperature is considered, as seen in Figure 2-6, a 5$^o$C rise in temperature was observed between days 10 – 21. This temperature increase coincided with the observed hotspots in figures 2-4a and 2-5a .With the subsequent reduction in the atmostpheric temperature consistently below 25$^0$C, after 70 days it was observed that, while some hotspots re-emerge their expanse and duration were reduced . The presence of temperatures above 25$^0$C were therefore interpreted as being a common factor impacting the oxygen profile in organic waste due to its importance in promoting microbial activity.

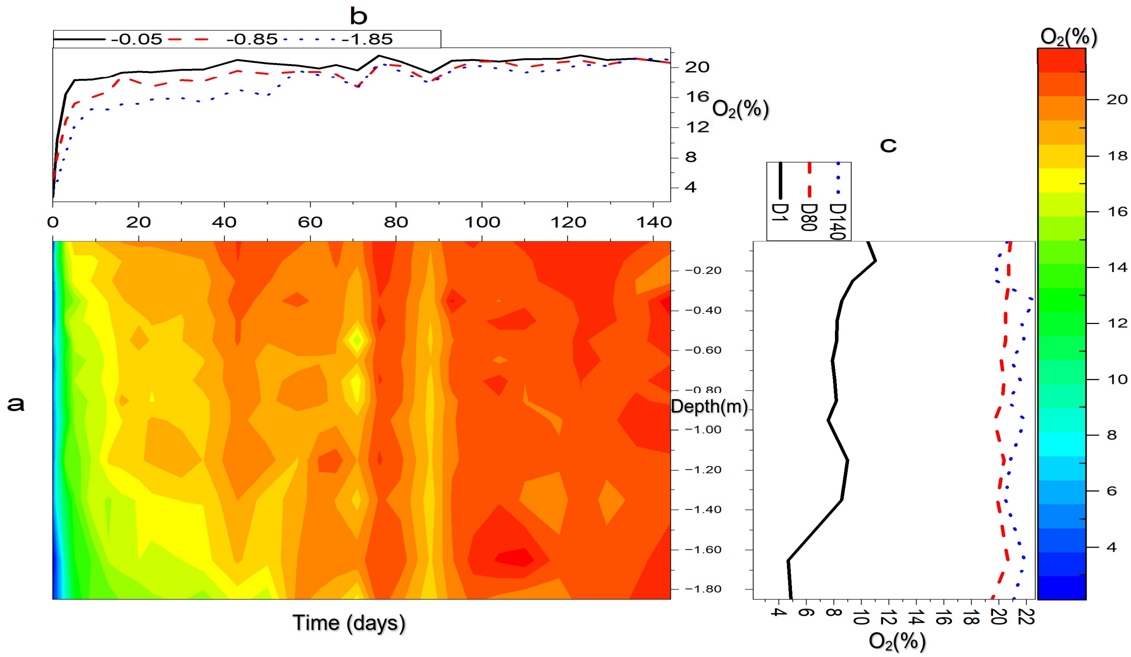


Figure 2-3. Oxygen Distribution 100%BA

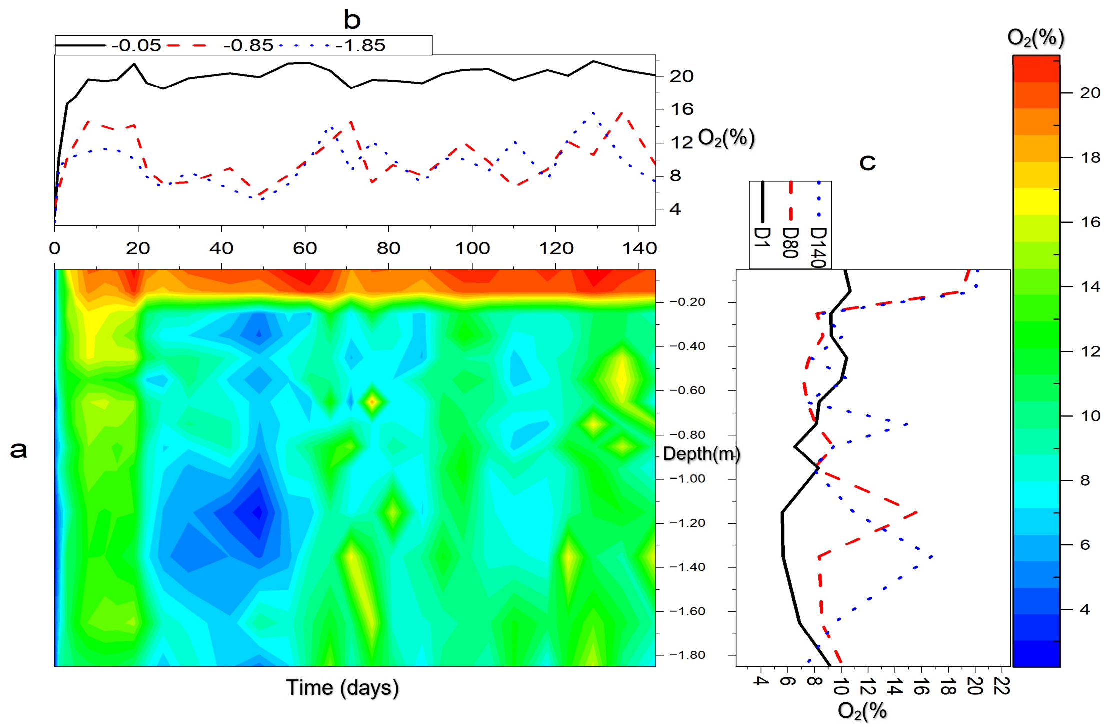


Figure 2-4. Oxygen Distribution 90%BA: 10% CP

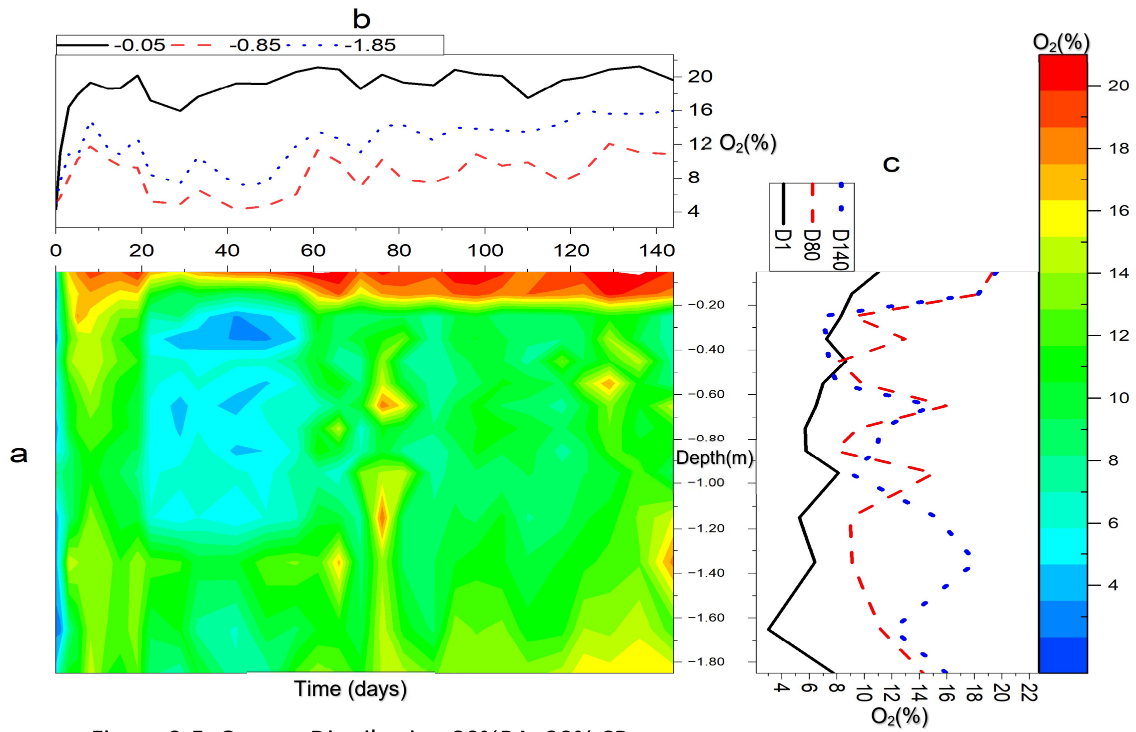


Figure 2-5. Oxygen Distribution 80%BA: 20% CP

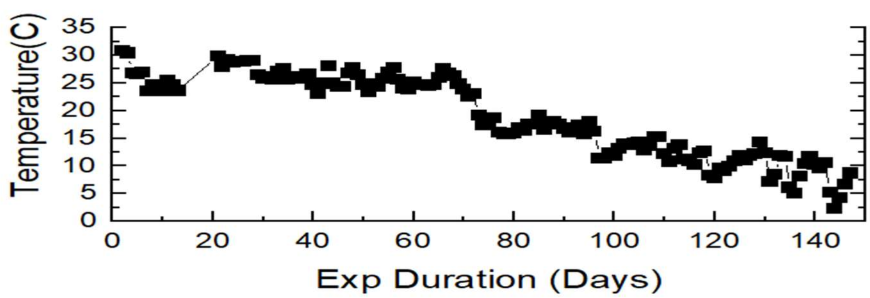


Figure 2-6. Atmospheric Temperature during experiment

### 2.3.3 Oxygen Consumption

From the oxygen distribution profiles the greatest oxygen debt, post column opening, was observed with 20%CP where the minimum oxygen level recorded was as low as 4.3%. However, this value is not significantly lower than the minimum recorded in the presence of 10% CP of 4.7%. Initial high oxygen consumption in low oxygen soils has been shown to be initially driven by oxidation of ions into oxides e.g., $Fe_2^+$, $NH_4^+$ combined with consumption by soil microorganisms (Reddy et al., 1980). The low oxygen levels observed with 10% CP and 20% CP after 20 days compared to the absence of additional organic matter shows the significant role of microorganisms in the consumption of oxygen in waste containing readily available substrates. Four separate areas, spanning the expanse of the columns, were selected as the points for the determination of maximal oxygen consumption rate. The rate was calculated utilizing the Crank–Nicolson form of the diffusion differential equation (Kallel et al., 2006)

$$\frac{\partial x}{\partial t} = \alpha \frac{\partial^2 x}{\partial k^2} \longrightarrow \frac{(x)_k^{t+\Delta t}-(x)_k^t}{\Delta t} = \frac{1}{2}\frac{D}{\xi}\left( \frac{(x)_{k-1}^t-2(x)_k^t+(x)_{k+1}^t}{\Delta y^2} + \frac{(x)_{k-1}^{t+\Delta t}-2(x)_k^{t+\Delta t}+(x)_{k+1}^{t+\Delta t}}{\Delta y^2} \right) - \rho_d \frac{RT}{\varepsilon P} R_1{}^t \quad \text{(Eq 2-1)}$$

Where x is measured oxygen mole fraction *(–)*, t is elapsed experimental time (days), k is sampling depth (m), D is the diffusion coefficient of oxygen ($m^2$/day), ξ is tortuosity*(–),* y is the distance between sampling points(m), P is the total pressure (Pa) , R is the universal gas constant ($m^3$ Pa $mol^{-1}$ $K^{-1}$), $\rho_d$ is the dry density T is temperature (K) and $R_1$ is the oxygen consumption rate.

Due to the discontinuity of the data, interpolation was conducted so that y=0.1m. Utilizing the Crank–Nicolson equation on this dataset resulted in days where there was no calculated positive oxygen consumption. Only days where net oxygen consumption was positive, were plotted and presented in Figure 6-9. These Figure show the change in the calculated oxygen consumption rate with time with the first Figure representing the consumption profile close to the soil-atmosphere boundary at -0.35m depth with the last Figure representing the consumption at the deepest point. In terms of the depth calculated consumption rates determined at the hotspot near to the 0.85M displayed the clearest trend in all columns with highest consumption occurring during the initial periods with a gradual decrease with time. This trend was consistent to that observed by Kallel et al. in previously landfilled wastes. However, this trend was not clearly discernable at each depth analyzed. In terms of the time the high scatter of the calculated oxygen consumption rates as well as low predictability of the consumption rate with time given by the low $R^2$ values of the line functions derived further demonstrate the highly variable nature of oxygen consumption in organic waste in the near term. Longer term investigations are thus needed to determine the oxygen consumption function of highly organic waste.

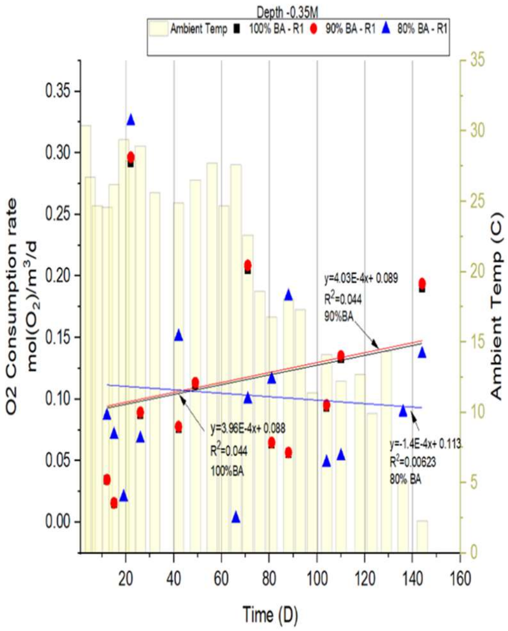


Figure 2-7: Calculated Oxygen Consumption Rate(-0.35M)

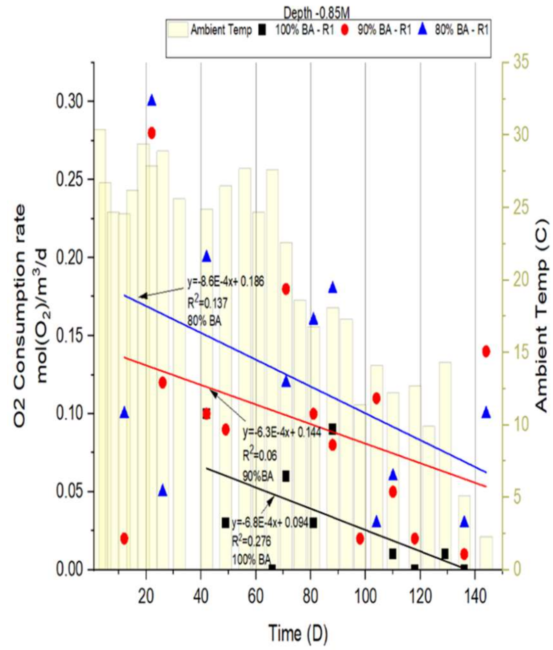


Figure 2-8: Calculated Oxygen Consumption Rate(-0.85M)

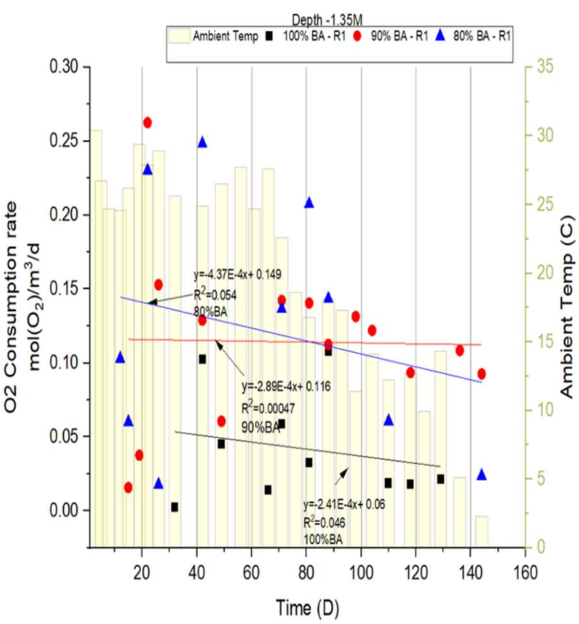


Figure 2-9: Calculated Oxygen Consumption Rate(-1.35M)

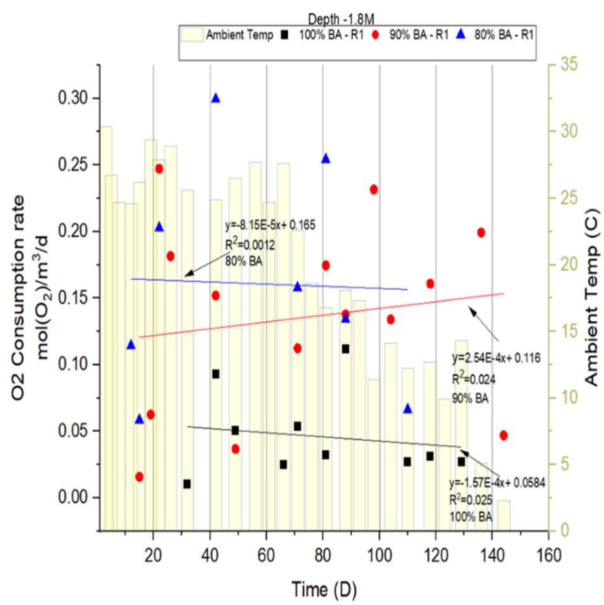


Figure 2-10: Calculated Oxygen Consumption Rate(-1.8M)

## 2.4 Conclusion

Landfills are by nature very heterogeneous masses, and this is reflected in the tendency of respiration hotspots developing. These areas are characterized as ‘pockets’ where oxygen is utilized at accelerated rates compared to the surrounding areas due to favorable conditions being present for bacterial proliferation. This results in a variable oxygen profile where the oxygen concentration, at any given depth, is not simply and accurately predicted as a function of time in the short term. It may be therefore more applicable to utilize generalized consumption rates for mature and stabilized wastes with high organic content to account for hotspot activity in areas with subsurface temperatures averaging above 20$^{o}$C. Further experimentation over an extended period at constant temperatures is recommended to create a general oxygen consumption function for organic wastes.

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# CHAPTER 3

## EFFECT OF VERTICAL GAS VENTILATION PIPES ON ORGANIC WASTE LANDFILL ASSESSED BY TWO-DIMENSIONAL LYSIMETERS

# CHAPTER 3

## 3. Effect of vertical gas ventilation pipes on organic waste landfill assessed by two-dimensional lysimeters.

3.1 INTRODUCTION

3.2 EXPERIMENTAL SETUP

3.3 RESULTS AND DISCUSSION

3.4 CONCLUSIONS

### 3.1 Introduction

Landfilled municipal solid waste (MSW) undergoes transformations which has been grouped into 5 distinct stages. (Cossu et al., 2018; Letcher & Vallero, 2019). Phase I, hydrolysis or aerobic phase, begins from the initial waste placement into cells as it immediately undergoes aerobic decomposition utilizing residual oxygen contained within the pore spaces at the time of layering and covering. Next, transitory phases (II+III) begin as oxygen is gradually consumed by heterotrophic, aerobic microorganisms. Under anoxic conditions, the methane fermentation phase or anaerobic phase (phase IV), which is characterized by methanogens or methane forming bacteria dominance, begins. The final phase, V, termed the maturation phase, where the waste mass is stabilized, and methane production is significantly reduced.

The anaerobic phase of landfills is associated with the production of landfill gas (LFG), whose constitution is comprised of approximately 55-65% methane and 35-45% carbon dioxide (Christensen et al., 2020). Globally there has been an increased effort to minimize fugitive methane gas emissions due to their significant contribution to global warming. LFG, specifically methane, has value as an energy source but in cases where its collection is not feasible or viable, in-situ aeration tactics have been implemented to assist in minimizing methane emissions as a post closure remediation mechanism (Ritzkowski and Stegmann, 2013). This led to numerous studies (Ritzkowski & Stegmann, 2012, 2018) on active aeration concepts where air is forcibly introduced into landfills, consuming energy via injection (positive aeration) or suction (negative aeration). An alternative in-situ aeration concept which has not been as extensively researched is the use of passive air vents. Traditionally, passive vents have been installed at landfills as a means of venting LFG from the landfill to the atmosphere to reduce LFG accumulation within the landfill mass. These traditional shafts can be perforated along most of their length to allow LFG created at any depth entry. Modification of these vents to contain perforations solely at the lower end, concentrated delivery of air into the waste can be achieved. This movement is driven by negative pressures created as LFG escapes (Ritzkowski and Stegmann, 2012). On-site studies analyzing the effect of passive aeration vents as a remediation mechanism showed that vents can be effective in delivering air to the lower regions of a landfill body if there is organic matter present (Kim et al., 2010). Kim et al. (2010) deduced that the vents facilitated aerobic degradation of organic matter which induced a temperature increase which drove convection within the waste leading to LFG rising and exiting while air entered. This gas movement created by the presence of standalone vertical vents mirrors the effect created by the semi aerobic method of landfilling. In the semi aerobic method, natural ventilation of waste is promoted via enlarged leachate collection pipes connected to vertical aeration pipes. Gas generated is removed from the waste mass via convection, driven by the residual heat from decomposition which allows for ambient air to enter via the leachate collection pipes promoting rapid stabilization of waste (Matsuto et al., 2015). Globally it is estimated that approximately 37% of waste is treated using landfills with only 8% of these landfills containing gas collection systems (Kaza et al., 2018). Of the remaining waste up to 33% is disposed of in open dumps, predominantly in low income / developing countries. To minimize impact on the natural environment these existing anaerobic and uncontrolled landfills require rehabilitation methods which can be implemented during the operating or post closure phases. With most of these sites located in low-income countries priority should be placed on rehabilitation which is low cost and simple to implement.

Passive aeration vents can be considered as an alternative, low cost, in situ landfill rehabilitation method which can be suited to rehabilitation of landfills in low income / developing countries. The study outline here aims to help clarify the applicability of vertical ventilation pipe installation at landfill sites in developing countries as a means of landfill improvement and or rehabilitation. This was assessed by visualizing the effect on gas distribution within a waste mass that occurs in the presence of stand-alone vertical gas ventilation pipes utilizing 2D lysimeters. Other investigations such as (Kim et al., 2010) were undertaken at situ at existing sites in the latter phases of the landfill lifecycle with depleted organic content. The use of lysimeters allowed for organic content to be standardized and for the investigation of the effect of vertical vents being implemented in waste of elevated organic content. During literature reviews no other investigations utilizing 2D lysimeters to investigate stand-alone passive aeration were discovered.

## 3.2 Experimental setup

### 3.2.1 2D Lysimeter

Three steel lysimeters were prepared with dimensions as shown in Figure 3-1 & 3-2. At the end of each lysimeter an S-trap pipe was attached to allow leachate to be collected while preventing air entry. The reverse side of each lysimeter contained sampling ports as pictured in Figure 3-1. The opposing face was composed of glass for observation of the filled contents as seen in Figure 3-3. S-trap pipes were attached to each lysimeter to ensure leachate release while simultaneously preventing air entry from below.

### 3.2.2 Ventilation Shaft

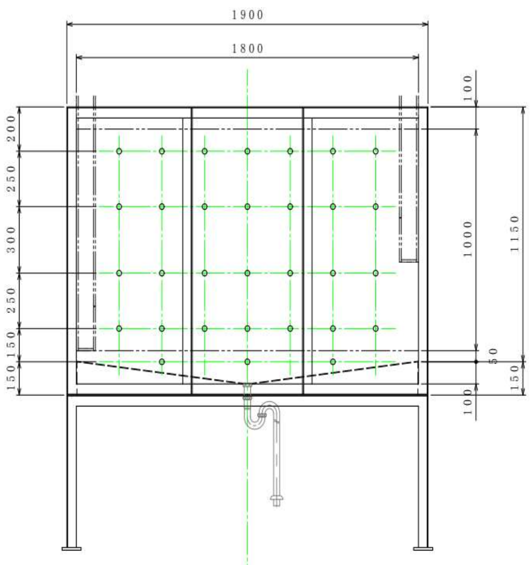
Two types of steel ventilation shafts were utilized: long and short. Long shafts were 1150mm in length and spanned from the drainage layer at its lowest point and extending to 50mm above the upper surface of the lysimeter at its highest point. The short shaft was 750mm in length. Apart from length, all shafts were of equal thickness and width. The final 200mm of each shaft were perforated with 10mm diameter openings as can be seen in Figure 3-5. The perforations were covered with a mesh to minimize clogging on insertion into the lysimeter.

### 3.2.3 Layering

Each lysimeter comprised three layers: a. Topsoil, b. Waste Layer, c. Drainage layer.

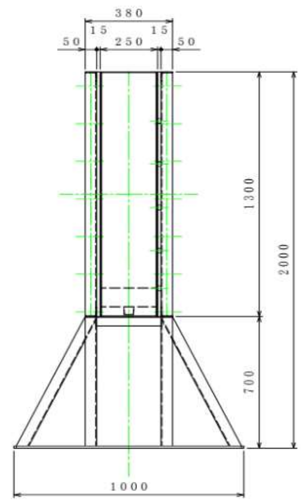
A. Topsoil –50mm in depth and constituted of inert sand-gravel mixture.

B. Waste layer- 1000mm in depth and composed of a mixture of 10% compost (CP) and 90% bottom ash (BA), on a dry weight basis. Compost was utilized to emulate the readily degradable organic




Unit: mm

Figure 3-1. 2D Lysimeter Rear View




Unit: mm

Figure 3-2. 2D Lysimeter Side View

Figure 3-3. Filled Lysimeter Front View- 2 Vents

Figure 3-4. Lysimeter Rear view showing gas sampling ports.

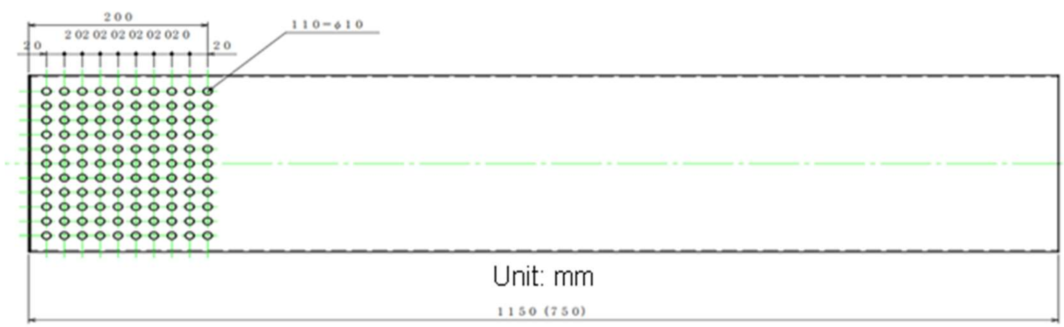



Figure 3-5. Ventilation Shaft

portion of waste while bottom ash was utilized to emulate the inorganic fraction. The waste layer was prepared by manually mixing the compost and bottom ash in the pre-determined ratio before adding it to the vessels in five layers. The layers were demarcated by the levels of gas ports. Each layer was uniformly compacted to attain a dry density of approximately 1200 kg/m$^3$.

C. Drainage layer- The drainage layer consisted of course stone of less than or equal to 50mm to facilitate leachate drainage and removal.

### 3.2.4 Controls

The first lysimeter (A) was established as a control with no ventilation shafts. The 2nd lysimeter (B) contained only a long shaft while the final lysimeter (C) contained both a long and short shaft placed on opposite sides of the lysimeter. Long shafts in lysimeters B and C were placed on the same side. No thermal insulation was implemented during the experimental 6-month period which extended from summer to the beginning of winter.

### 3.2.5 Gas Sampling

Thirty-one gas samples were obtained from the sampling ports connected to each lysimeter at weekly intervals. Samples were extracted utilizing a 5ml syringe and two 3-way stop valves; one valve was fixed to the sampling port of each lysimeter with the other attached to the sampling syringe. A maximum of 3ml of air was extracted for each sample. Gas samples were analysed for the presence $O_2$, $CO_2$, $CH_4$ and $N_2$ via SHIMADZU GC-2014 gas chromatograph.

### 3.2.6 Leachate Sampling

Leachate samples were collected on a fortnightly after the first month of the lysimeters being established. Samples were analyzed for the concentrations of cations ($Ca^{2+}$, $Na^+$, $K^+$, $Mg^{2+}$), anions ($Cl^-$,$NO_3^-$,$SO_4^{2-}$), Organics (TC, TOC), Nitrogen (TN, $NH_4^+$), pH, Oxidation Reduction Potential (ORP) and Conductivity (EC). Anions were analyzed by Dionex ICS 2100, Cations by Dionex ICS 1100 and TOC, TN using a Shimadzu TOC-V CPH Analyzer. pH and EC were analyzed using Horiba Laqua F47BW while ORP was tested using the Horiba Navi F-51.

## 3.3 Results and discussion

### 3.3.1 Waste layer

The organic content of the waste layer materials was determined prior to mixing and placement in lysimeters. The loss on ignition (LOI) method, which is one of the most frequently utilized means of determining organic content in soils, was utilized. Samples were oven dried at 200°C for 24hrs. The difference between the weight before drying and that after drying was interpreted as the weight loss due to moisture. LOI was determined by ignition of the oven dried samples in a muffle furnace for 2 hours at 600°C. The difference between the weight before and after ignition was interpreted as the weight loss due to organic matter. Organic matter was converted to organic carbon utilizing a conversion factor of 0.55 $kgkg^{-1}$. (Hoogsteen et al., 2015).

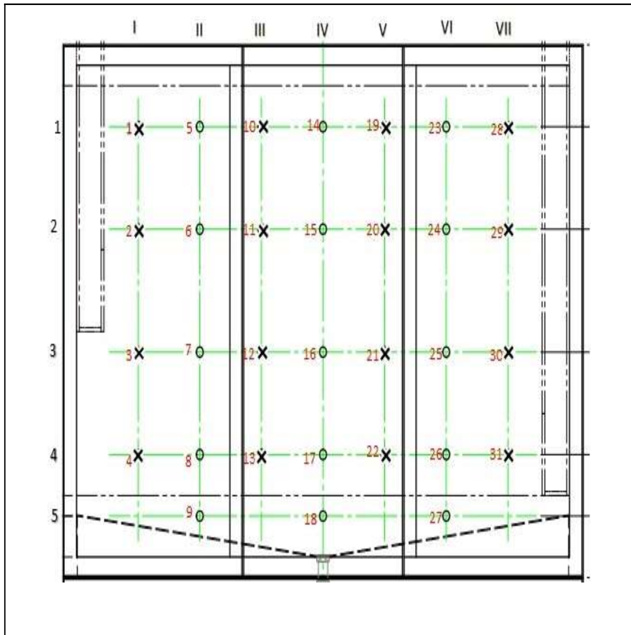
Table 3-I presents the result of LOI determination. CP utilized in the experiment contained, on average, 44% organic carbon, with BA containing insignificant quantities. Utilizing this data together with the lysimeter filling ratio of 10%CP: 90% BA and the filling density of 1200kg/$m^3$ the available carbon was estimated at 56kg/$m^3$ or 5% of the mass of each lysimeter.

Table 3-I Waste Layer Organic Content

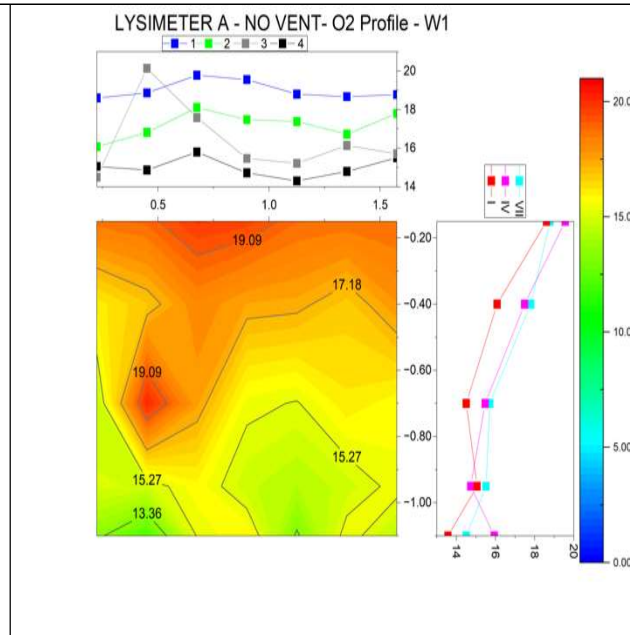
| BOTTOM ASH | | | |
|---|---|---|---|
| Sample # | Moisture Content (%) | Mass loss on ignition (% ) | Estimated Organic Carbon (%) |
| 1 | 25.7 | 0.0 | 0.00 |
| 2 | 26.4 | 1.3 | 0.71 |
| 3 | 26.9 | 0.0 | 0.00 |
| 4 | 25.2 | 1.3 | 0.71 |
| 5 | 23.3 | 0.0 | 0.00 |
| 6 | 23.8 | 1.3 | 0.71 |
| Mean | 25.21 | 0.65 | 0.355 |
| COMPOST | | | |
| Sample # | Moisture Content (%) | Mass loss on ignition (% ) | Estimated Organic Carbon (%) |
| 1 | 13.5 | 78.9 | 43.39 |
| 2 | 14.3 | 80 | 44.00 |
| 3 | 14 | 79.1 | 43.49 |
| 4 | 1 | 79.8 | 43.89 |
| 5 | 13.2 | 82.6 | 45.43 |
| 6 | 12.9 | 80.7 | 44.38 |
| Mean | 11.48 | 80.18 | 44.09 |

### 3.3.2 Gas distribution

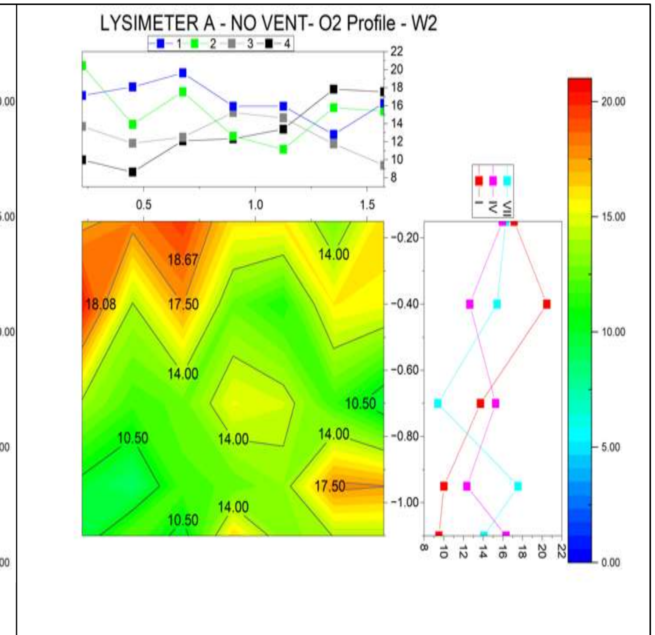
Th The utilization of 2D lysimeters allowed for the determination of both the vertical and lateral oxygen distribution profiles. These gas profiles were visually rendered as gas contour graphs where zones which were not directly measured were interpolated. Figure 3-6 to 3-8 present the oxygen contour maps for each lysimeter in four-week intervals where the right of the Figure represents the location of long vertical vents (Figure 3-7& 3-8) and the left being the location of the short vent (Figure 3-8)



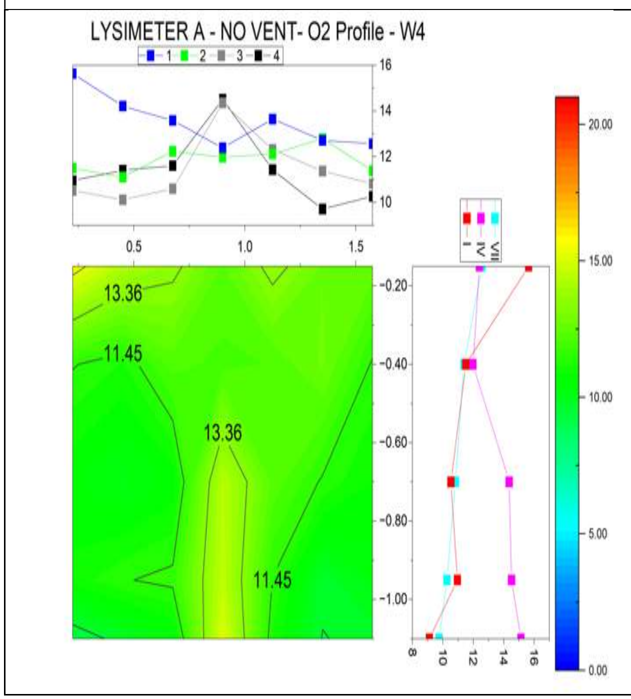
I II III IV V VI VII
1 2 3 4 5
1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23 24 25 26 27 28 29 30 31

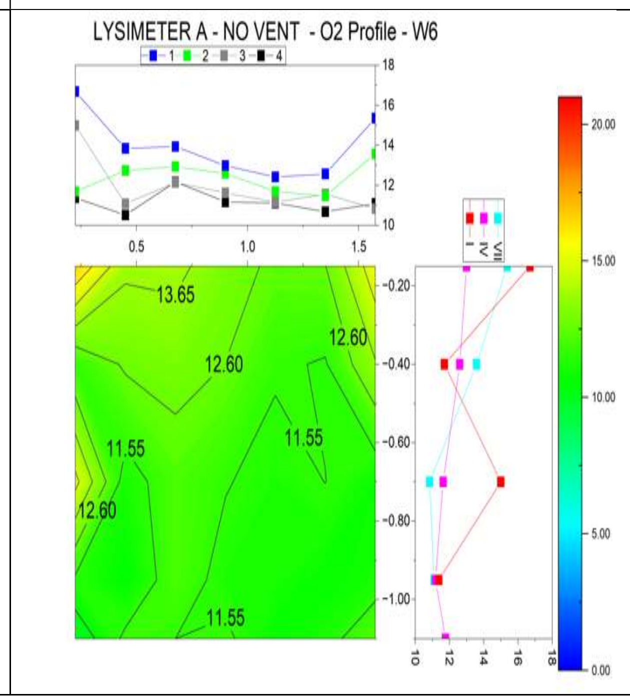
LYSIMETER A - NO VENT- O2 Profile - W1
1 2 3 4
I IV VII
19.09
17.18
19.09
15.27
15.27
13.36

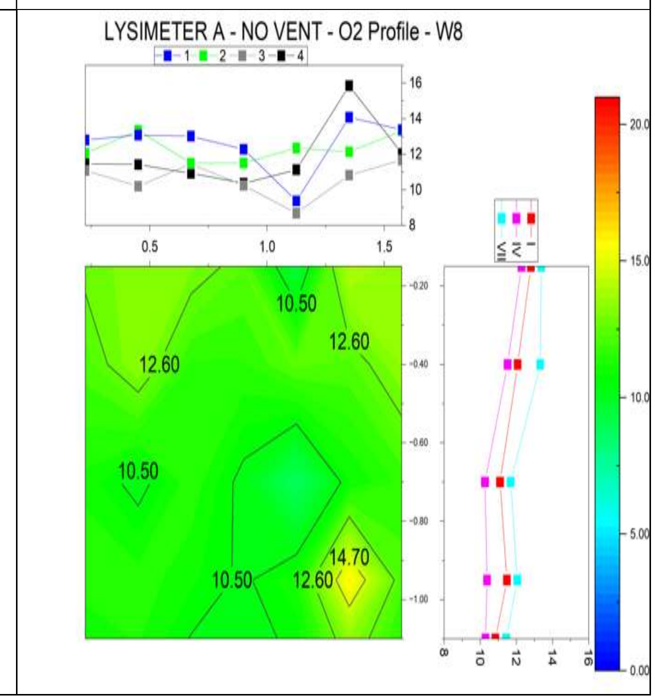
LYSIMETER A - NO VENT- O2 Profile - W2
1 2 3 4
I IV VII
18.67
14.00
18.08
17.50
14.00
10.50
10.50
14.00
14.00
17.50
14.00
10.50

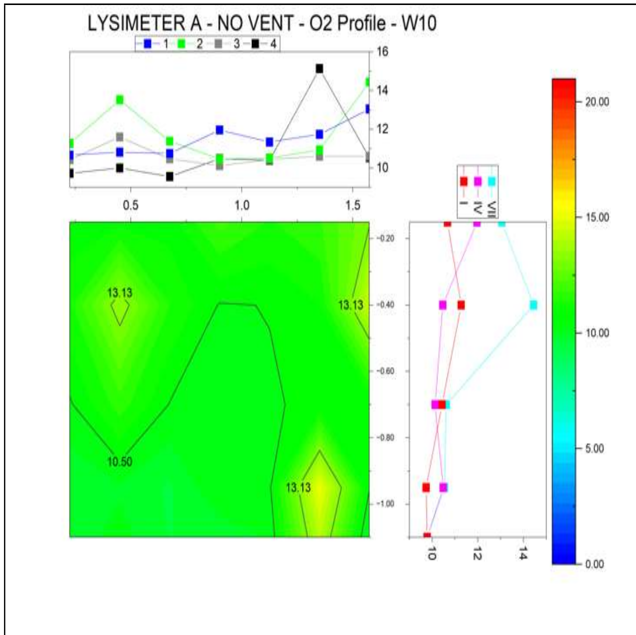
LYSIMETER A - NO VENT- O2 Profile - W4
1 2 3 4
I IV VII
13.36
11.45
13.36
11.45

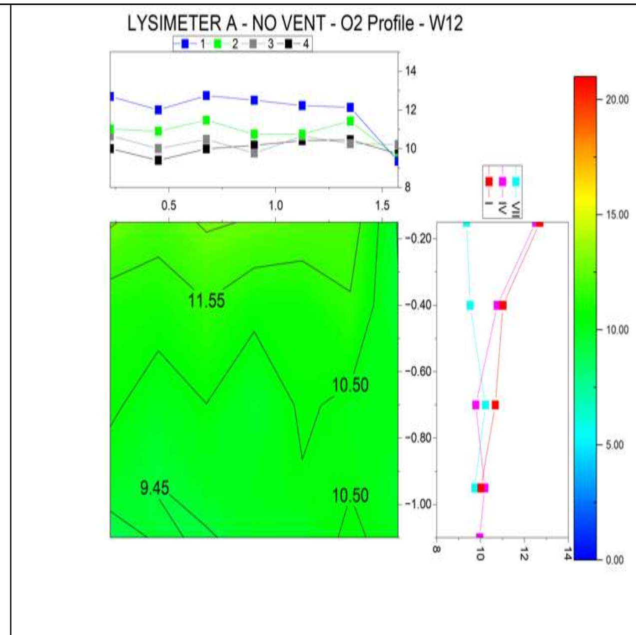



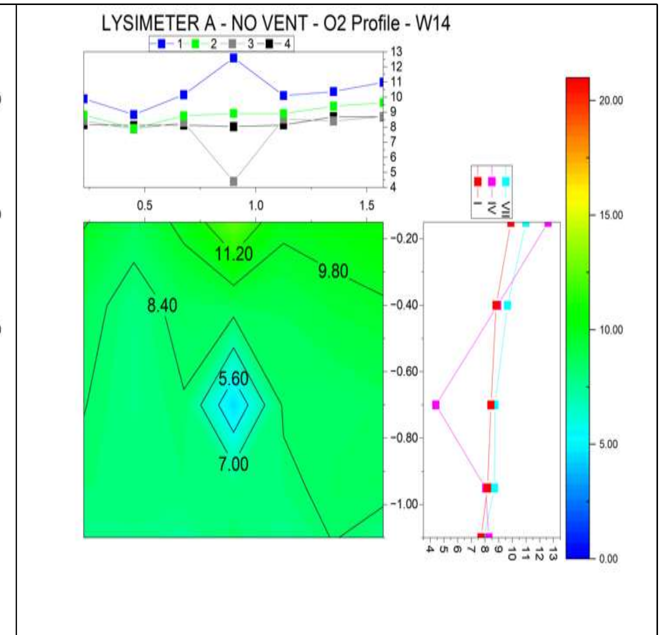
LYSIMETER A - NO VENT - O2 Profile - W6
1 2 3 4
I IV VII
13.65
12.60
12.60
11.55
11.55
12.60
11.55

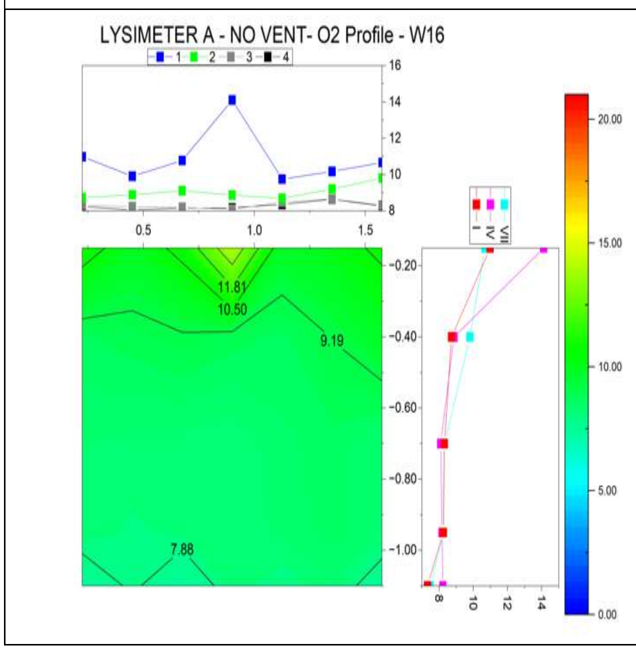



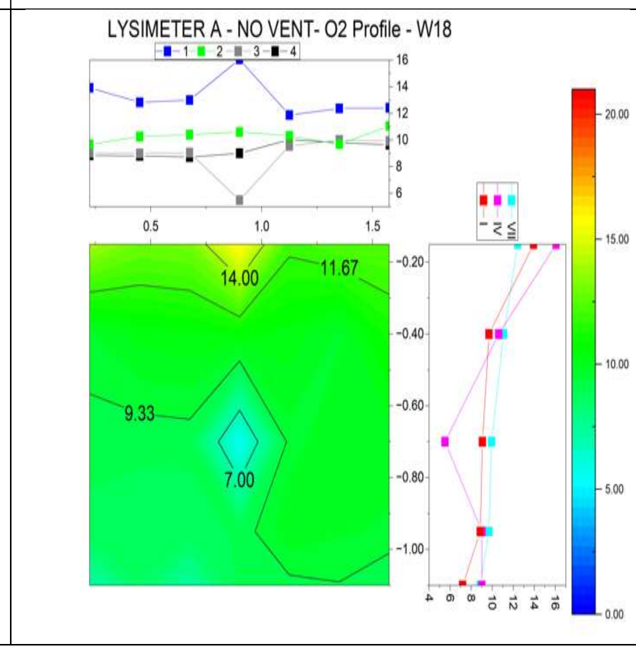
LYSIMETER A - NO VENT - O2 Profile - W8
1 2 3 4
I IV VII
10.50
12.60
12.60
10.50
14.70
10.50
12.60

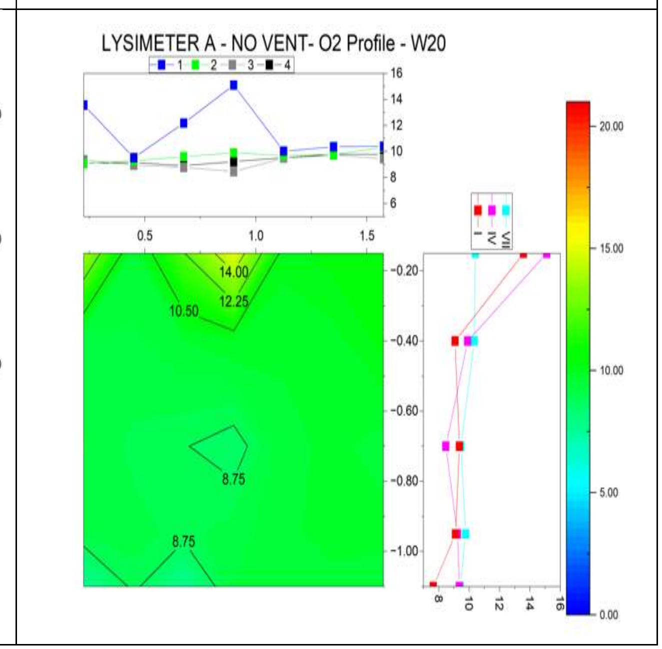

LYSIMETER A - NO VENT - O2 Profile - W10
1 2 3 4
I IV VII
13.13
13.13
10.50
13.13


LYSIMETER A - NO VENT - O2 Profile - W12
1 2 3 4
I IV VII
11.55
10.50
9.45
10.50


LYSIMETER A - NO VENT - O2 Profile - W14
1 2 3 4
I IV VII
11.20
9.80
8.40
5.60
7.00


LYSIMETER A - NO VENT- O2 Profile - W16
1 2 3 4
I IV VII
11.81
10.50
9.19
7.88


LYSIMETER A - NO VENT- O2 Profile - W18
1 2 3 4
I IV VII
14.00
11.67
9.33
7.00


LYSIMETER A - NO VENT- O2 Profile - W20
1 2 3 4
I IV VII
14.00
12.25
10.50
8.75
8.75

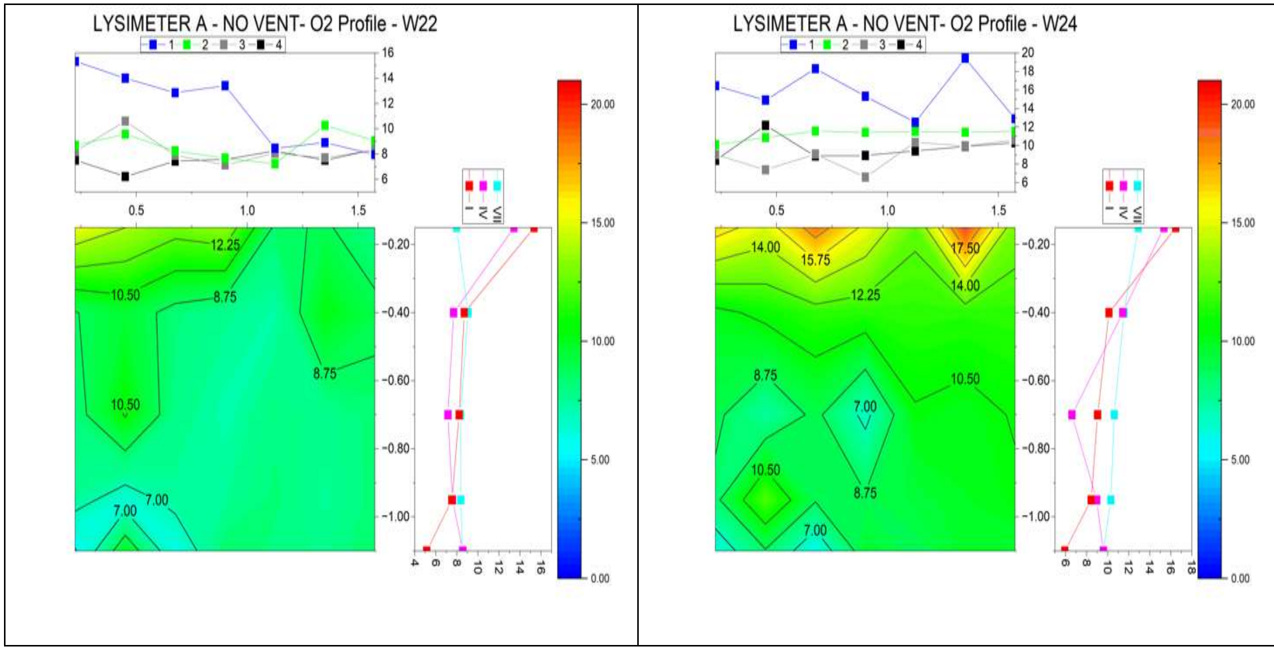
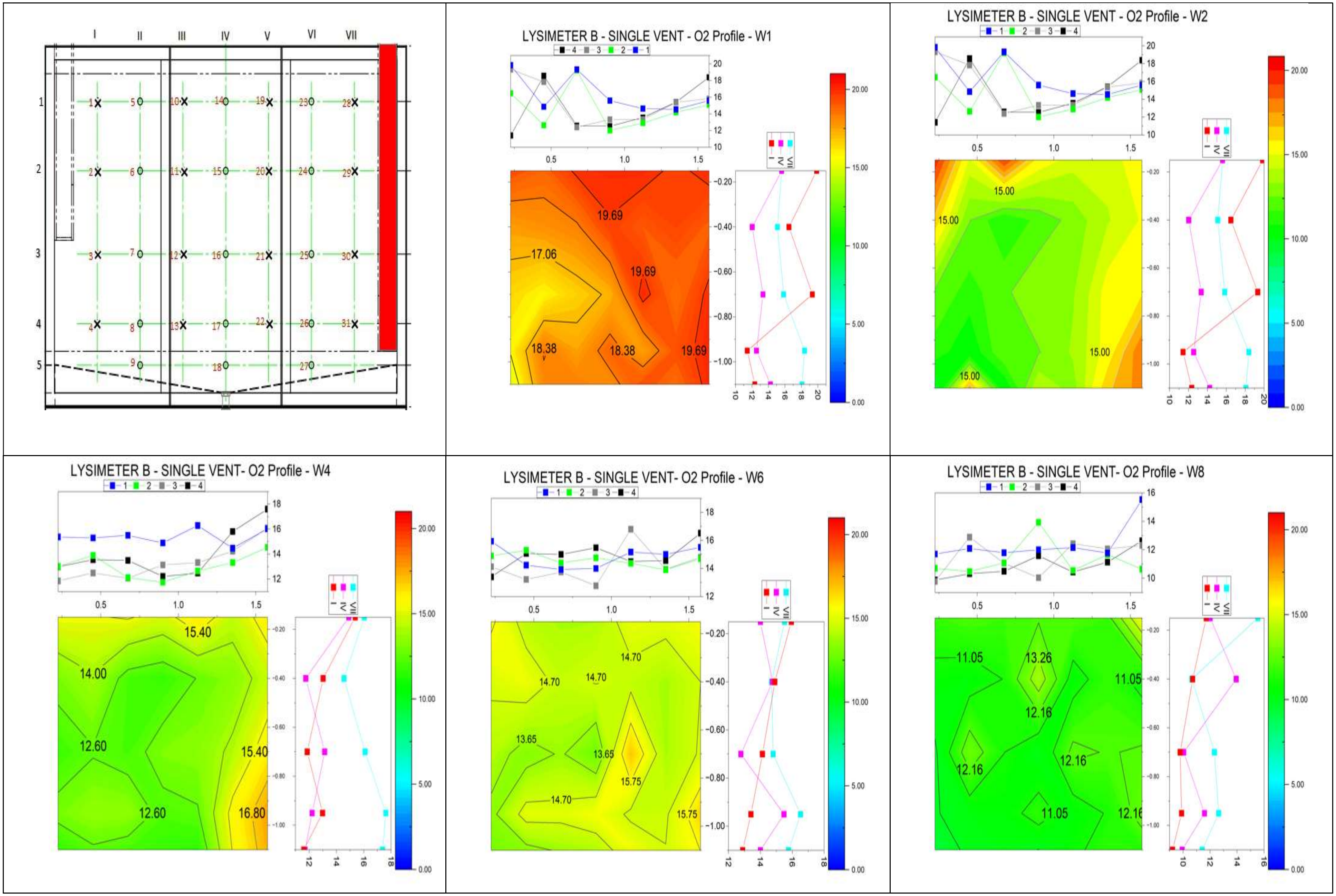
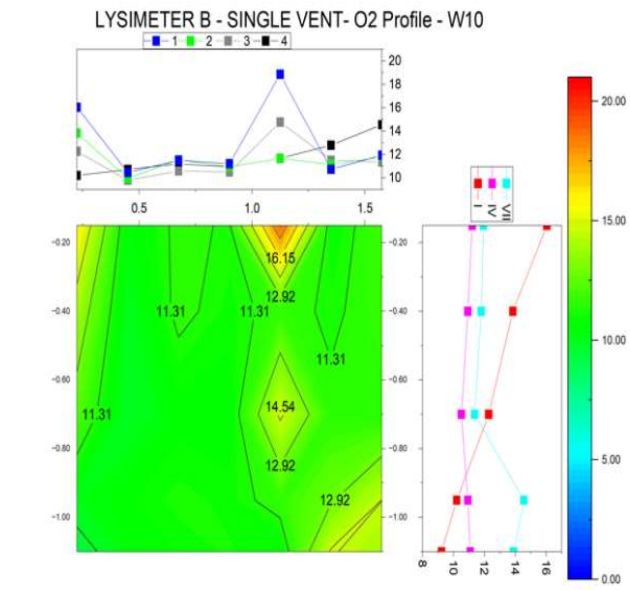
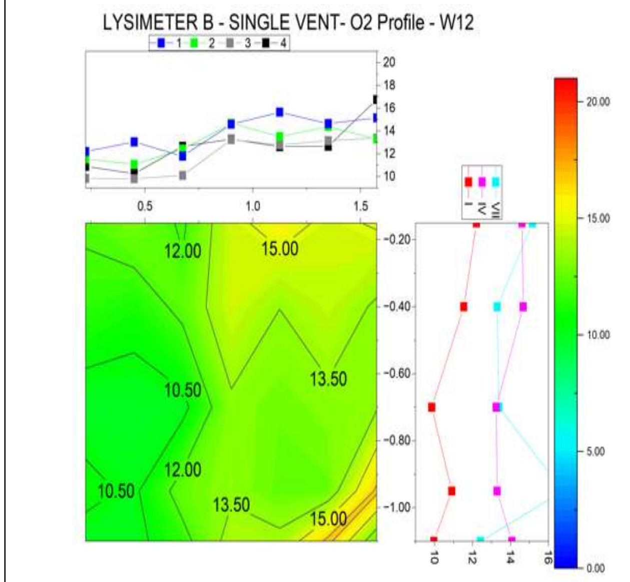
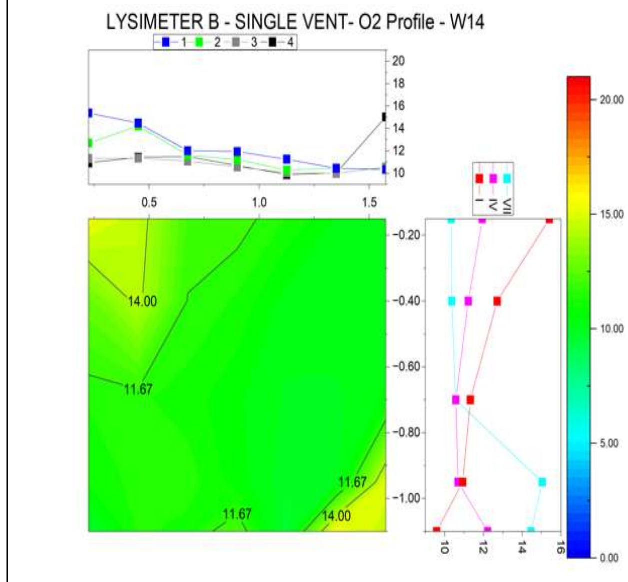


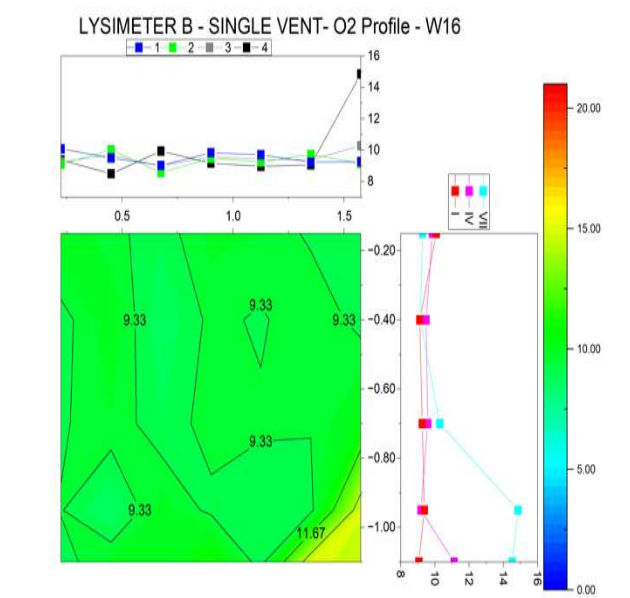

Figure 3-6 No Vents - $O_2$(%) distribution

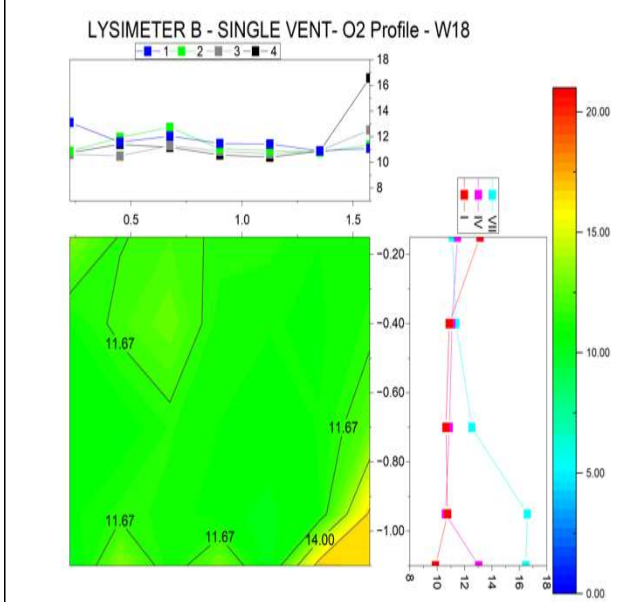

I II III IV V VI VII
1 2 3 4 5
LYSIMETER B - SINGLE VENT - O2 Profile - W1
19.69
17.06
18.38
LYSIMETER B - SINGLE VENT - O2 Profile - W2
15.00
LYSIMETER B - SINGLE VENT- O2 Profile - W4
15.40
14.00
12.60
16.80
LYSIMETER B - SINGLE VENT- O2 Profile - W6
14.70
13.65
15.75
LYSIMETER B - SINGLE VENT- O2 Profile - W8
11.05
13.26
12.16

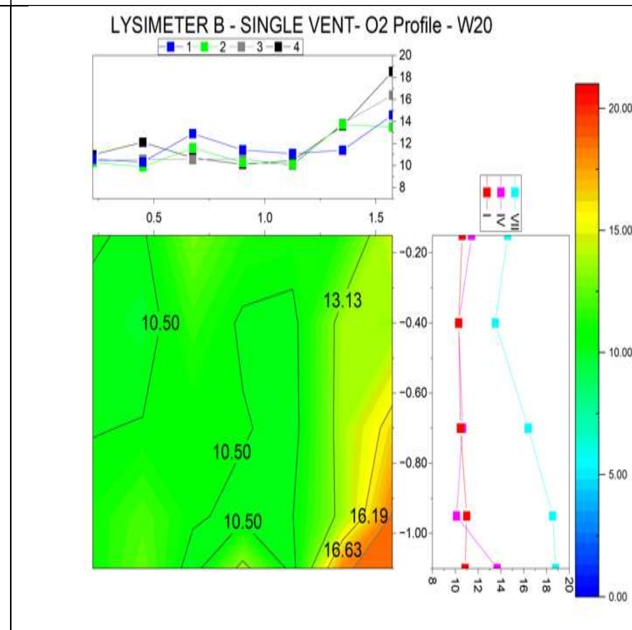

LYSIMETER B - SINGLE VENT- O2 Profile - W10

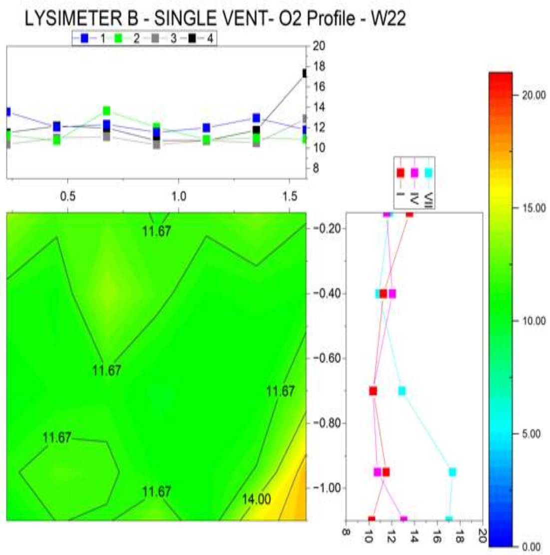

LYSIMETER B - SINGLE VENT- O2 Profile - W12


LYSIMETER B - SINGLE VENT- O2 Profile - W14


LYSIMETER B - SINGLE VENT- O2 Profile - W16


LYSIMETER B - SINGLE VENT- O2 Profile - W18


LYSIMETER B - SINGLE VENT- O2 Profile - W20



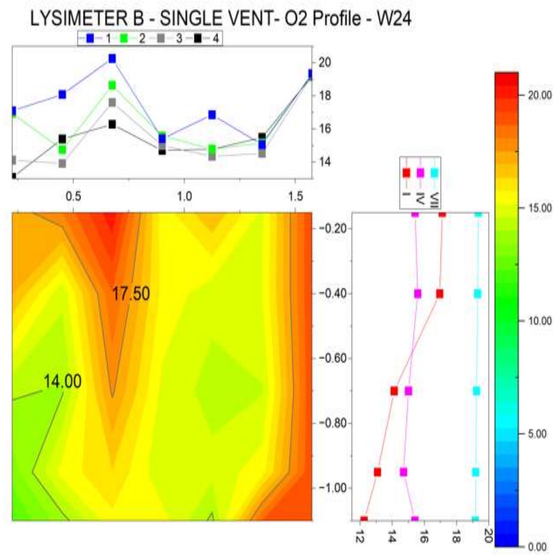
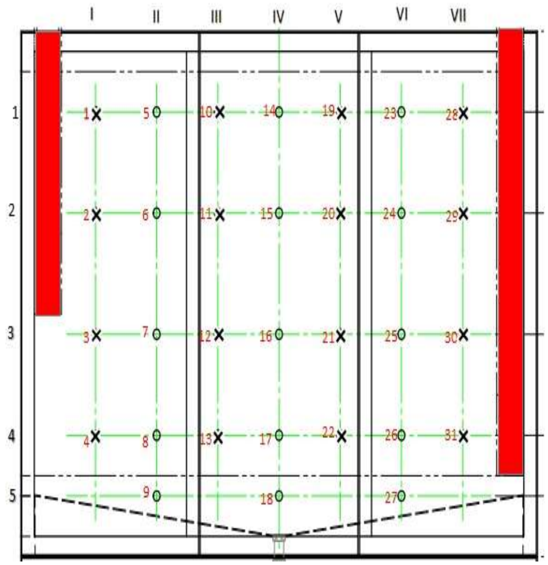
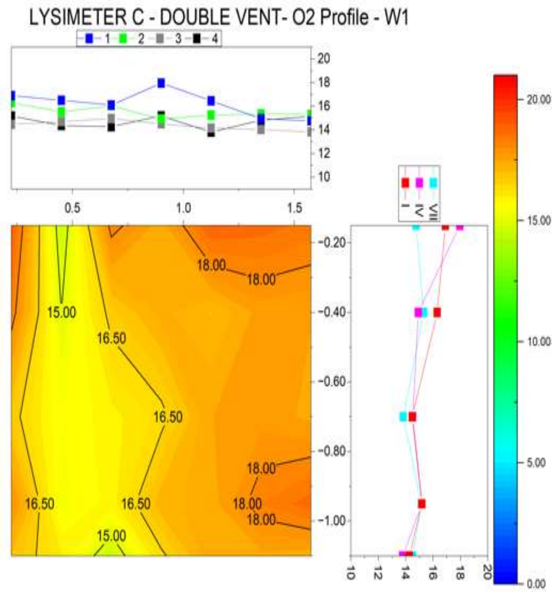
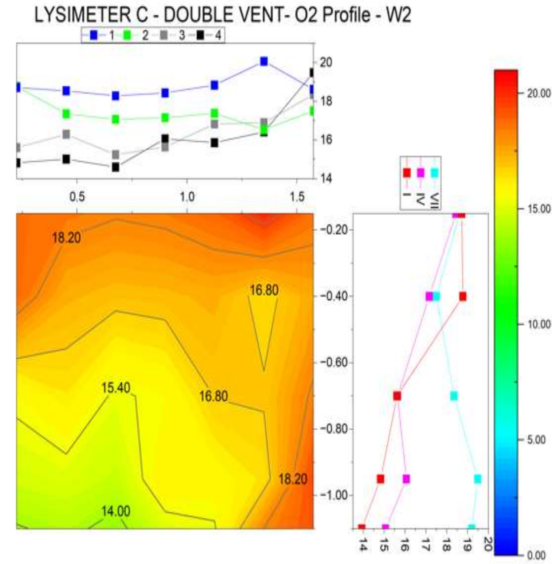
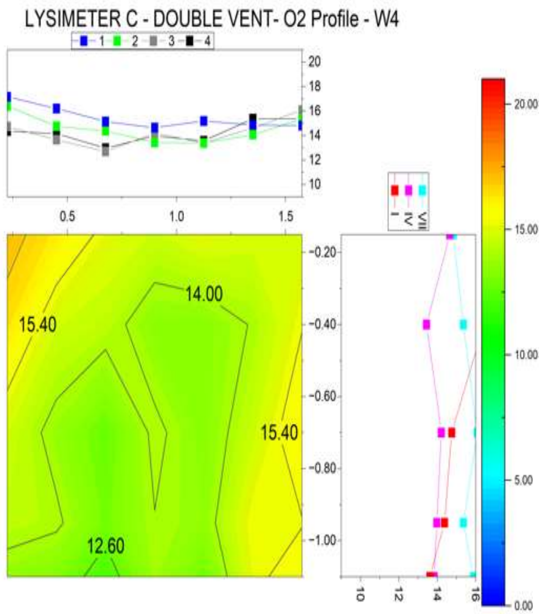



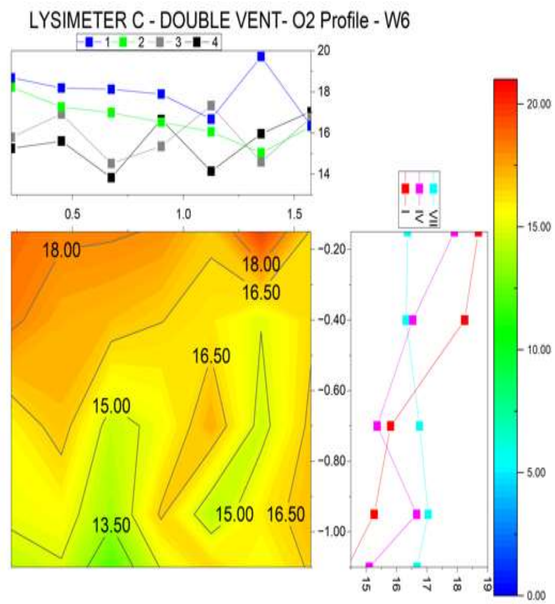

Figure 3-7 Single Long Vent (Right) - $O_2$(%) distribution

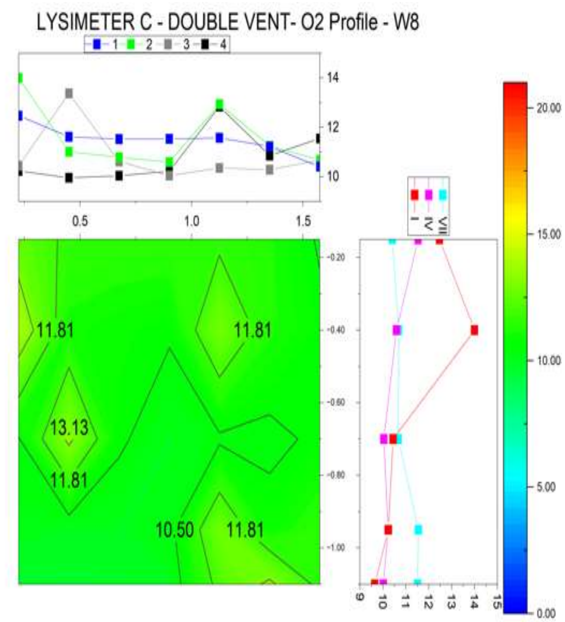


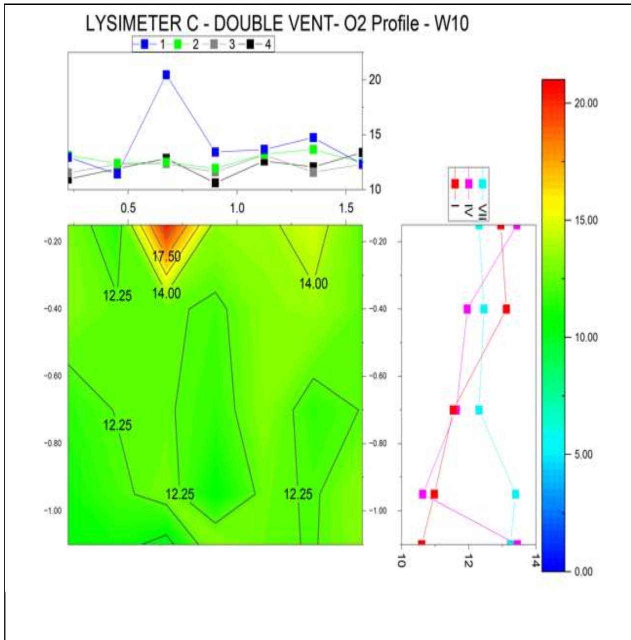

LYSIMETER C - DOUBLE VENT- O2 Profile - W1

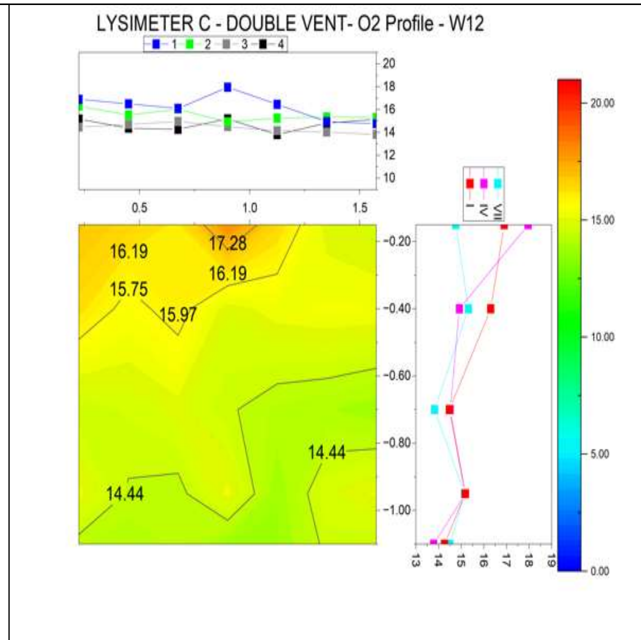

LYSIMETER C - DOUBLE VENT- O2 Profile - W2

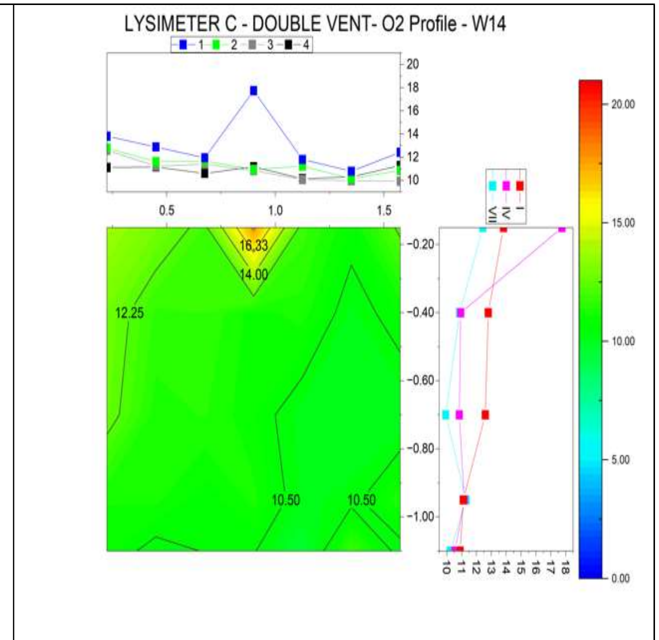

LYSIMETER C - DOUBLE VENT- O2 Profile - W4

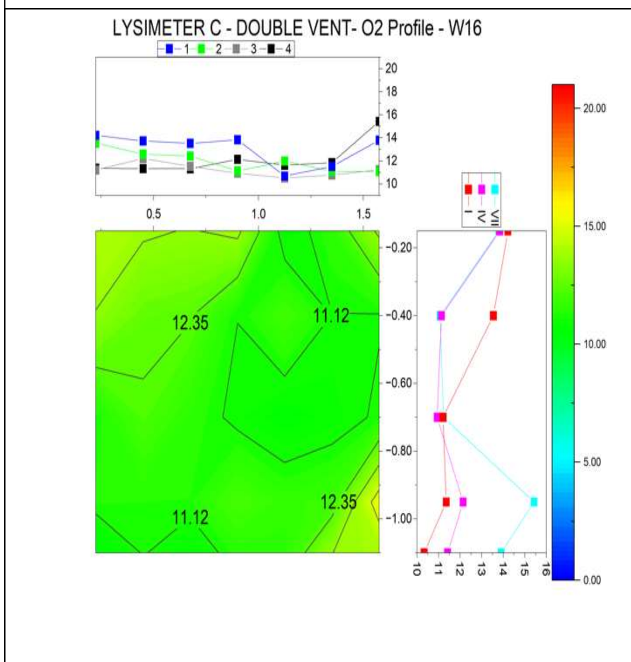

LYSIMETER C - DOUBLE VENT- O2 Profile - W6

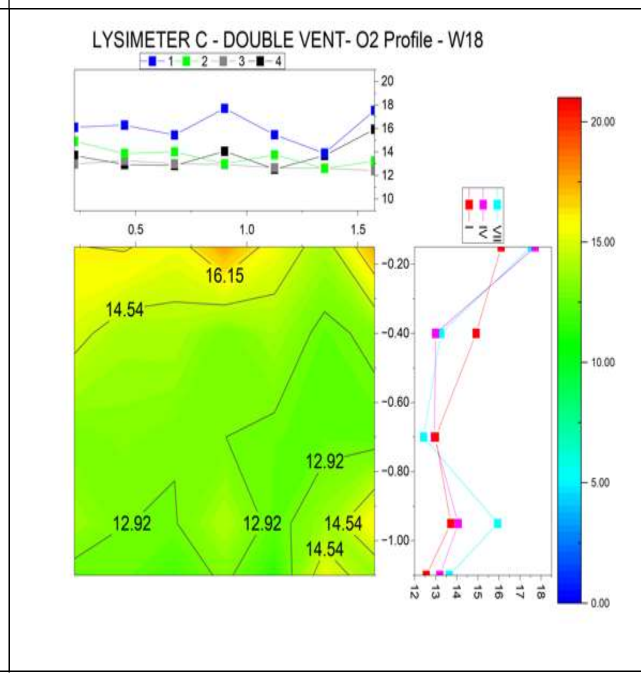

LYSIMETER C - DOUBLE VENT- O2 Profile - W8

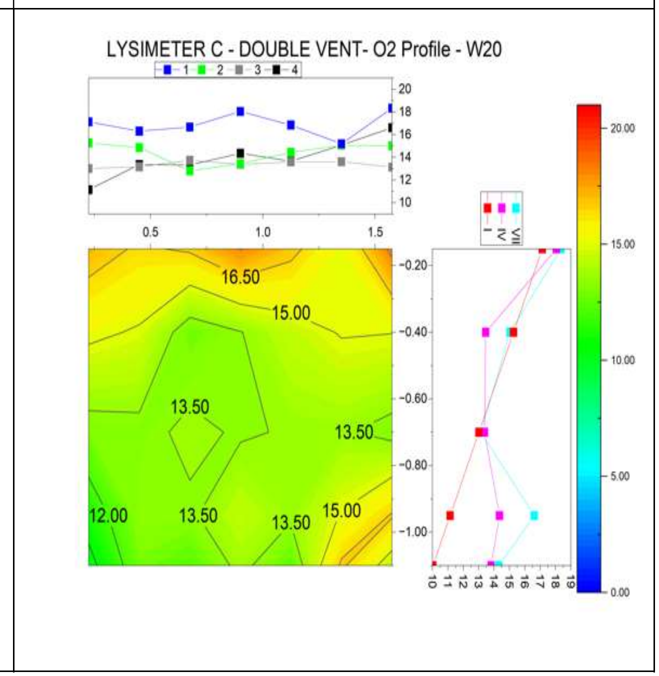

LYSIMETER C - DOUBLE VENT- O2 Profile - W10
1 2 3 4
I IV VII


LYSIMETER C - DOUBLE VENT- O2 Profile - W12
1 2 3 4
I IV VII


LYSIMETER C - DOUBLE VENT- O2 Profile - W14
1 2 3 4
I IV VII


LYSIMETER C - DOUBLE VENT- O2 Profile - W16
1 2 3 4
I IV VII


LYSIMETER C - DOUBLE VENT- O2 Profile - W18
1 2 3 4
I IV VII


LYSIMETER C - DOUBLE VENT- O2 Profile - W20
1 2 3 4
I IV VII

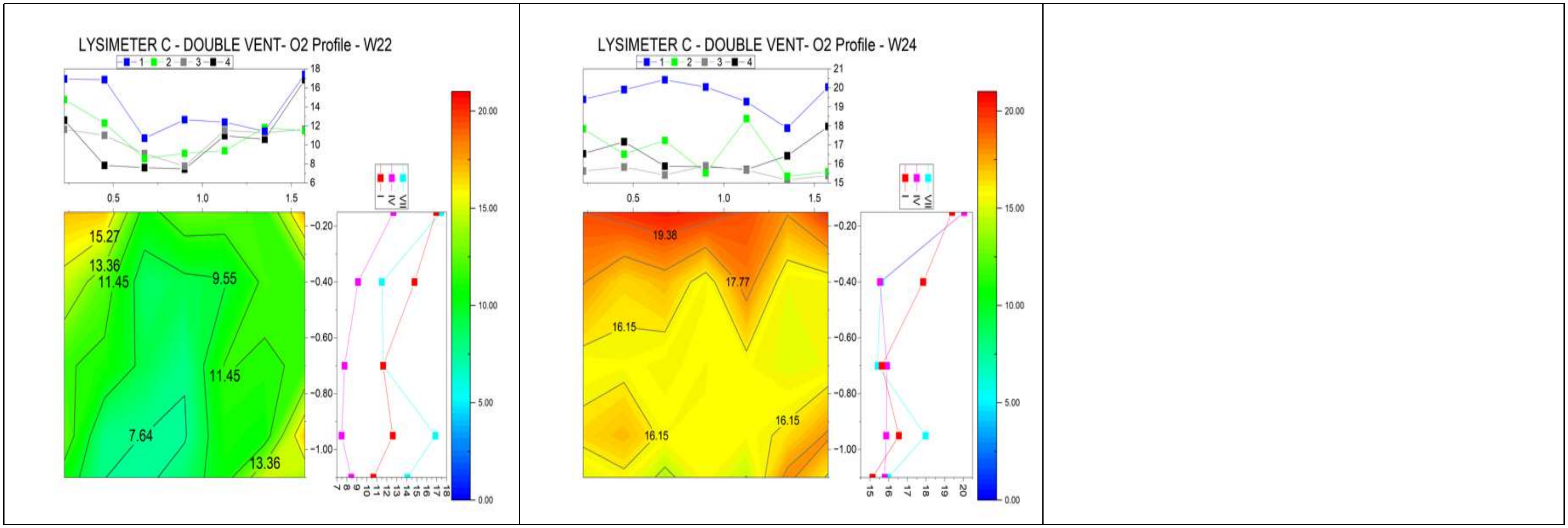


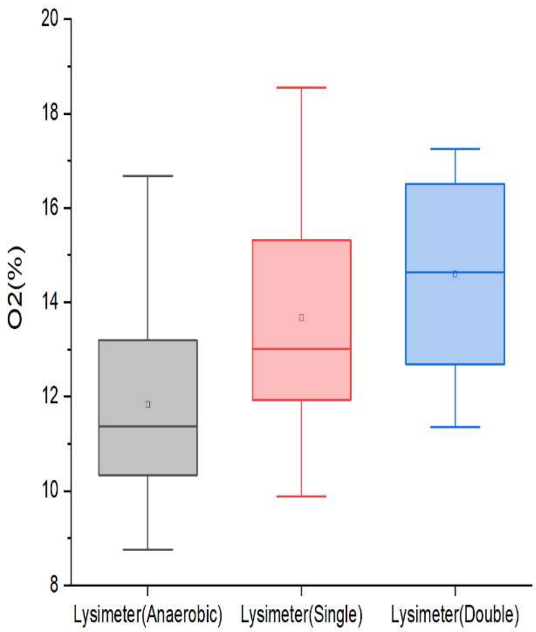

Figure 3-8 Short Vent (Left) Long Vent (Right)- $O_2$(%) distribution.

When waste is initially placed, oxygen is trapped in pore spaces which allows for existing populations of aerobic bacteria in compost to proliferate utilizing the moisture content available in the bottom ash and supplied manually. With readily available carbon sources, oxygen is gradually used up. This decline in oxygen levels was observed in all lysimeters as the maximum oxygen concentrations measured in the lower reaches of each lysimeter can be seen to gradually decrease. The decrease is however more pronounced where no gas ventilation shafts were present. With no ventilation shafts, oxygen diffuses solely from the surface which results in a noticeable concentration gradient visible in the contours observed in Figure 3-6. In the absence of ventilation pipes at 14 weeks respiratory hotspots can be observed which are described in the previous chapter. Long ventilation shafts were able to deliver oxygen preferentially to the lower regions of the waste mass as visible in Figure 3-7&3-8. At the lower right of these contour maps elevated oxygen levels were observed when compared to the levels measured at the centre of the waste mass. This region represents the location of the perforations of the long ventilation shafts. These shafts appear to have effective radius of between 0.5 -1 meters based on the analysis of the contour data of the single long ventilation shaft present in the lysimeter depicted in Figure 3-7. Here it was observed that beyond the 1-meter mark at the furthest corner away from both the surface and the ventilation shaft the oxygen concentrations were not much greater than those measured in the lysimeter where vents were absent.

The highest overall oxygen levels were persistently observed in the lysimeter containing two shafts. Air enters at both shafts delivering air into the waste mass thus creating a better distribution.

The reduction in oxygen concentration over time was greatest in the lysimeter A with a minimum level as low as 7% measured while minimums of approximately 9.5% & 10.5% measured in B and C respectively. This demonstrated that the presence of the shafts induces increased oxygen levels passively. Additional effects of the shafts were observed by looking at the lysimeter B where, in Figure 3-7, depths below 0.8m there were consistently higher oxygen concentrations, ranging from 1-5%, where the long shaft was positioned (right side). This effect was also observed in the lysimeter C.

To quantify the effect of the passive vents the mean oxygen concentration of each lysimeter for each sampling day was collated and subjected to a means test with the hypothesis that there was no difference between means. The summarized result for this analysis is presented in Figure 3-9. Using a significance level of 0.05 the analysis showed that, with a probability level of 0.000627 there is a significant difference in the mean oxygen levels between the lysimeter without vents (A) and the two with vents (B+C). Pairwise comparisons confirmed that lysimeters containing vents have a higher average $O_2$ concentration than that without, but the additional short vent did not produce a significant improvement compared to having only a single long vent. Lysimeter B averaged 13.7% $O_2$ while the additional short vent in lysimeter C only increased the average to 14.6%, a 6.7% increase. Comparison of Lysimeter's A and B showed a 15.7% increase in lysimeter B over A.




| | Mean | Standard Deviation | Standard Error of Mean | Probability | Tukey Assessment * |
|---|---|---|---|---|---|
| Lysimeter(A) | 11.82 | 2.17 | 0.48 | | Y |
| Lysimeter(B) | 13.67 | 2.32 | 0.51 | | Z |
| Lysimeter(C) | 14.59 | 1.99 | 0.44 | | Z |
| | | | | 6.27E-4 | |


| | Mean | Standard Deviation | Standard Error of Mean | Probability | Tukey Assessment * |
|---|---|---|---|---|---|
| Lysimeter(A) | 11.82 | 2.17 | 0.48 | | Y |
| Lysimeter(B) | 13.67 | 2.32 | 0.51 | | Z |
| Lysimeter(C) | 14.59 | 1.99 | 0.44 | | Z |
| | | | | 6.27E-4 | |


Figure 3-9. Mean $O_2$ concentration.

### 3.3.3 LEACHATE QUALITY

Leachate quality is closely correlated to the age and phase of decomposition of the waste mass (Kjeldsen et al., 2002). Typical leachate TOC concentrations ranging between 30-29,000mg/L were reported by Kjeldsen et al. (2002). As can be observed in Figure 3-10 the experimental lysimeters had initial concentrations ranging from 14-19,000 mg/L, near the higher end of the range reported by Kjeldsen et al. thus the organic load was comparable to landfills. Over the experimental period both ventilated lysimeters displayed lower TOC concentrations than the un-ventilated lysimeter. However, lysimeter B consistently displayed lower TOC levels compared to lysimeter C. Aeration experiments (Hrad and Huber-Humer, 2017; Slezak et al., 2015) typically show that higher oxygen levels created by aeration typically result in greater TOC reductions in leachate. It was therefore expected that lysimeter C with its higher absolute mean oxygen level would result in marginally better absolute TOC reduction. This did not prove to be the case with the

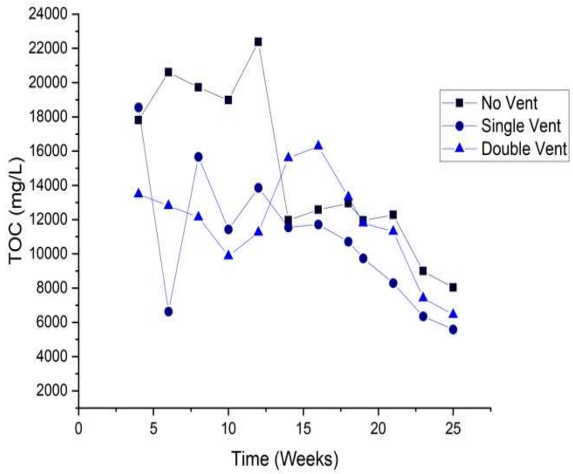



Figure 3-10. Leachate Total Organic Carbon (TOC)

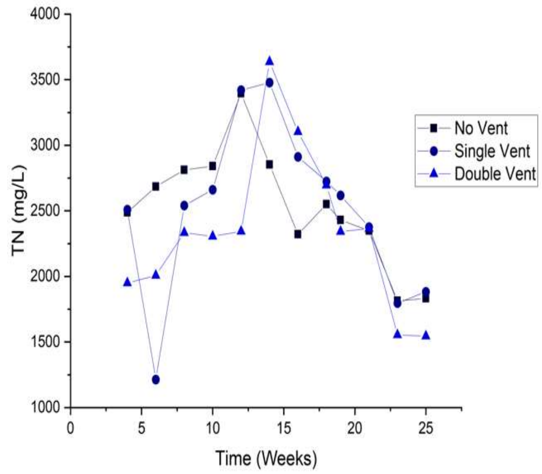



Figure 3-11. Leachate Total Nitrogen (TN)

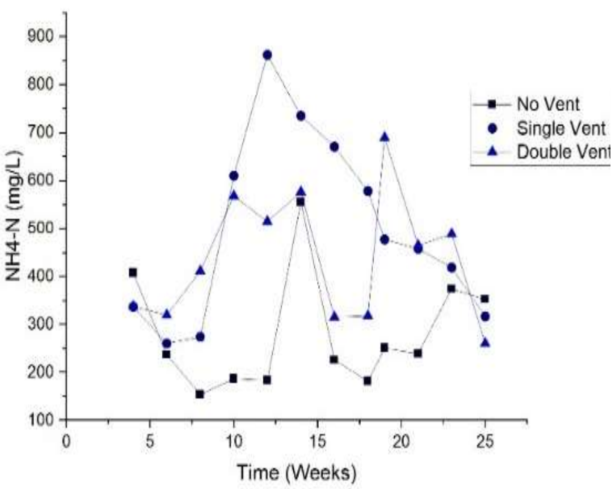



Figure 3-12. Leachate Ammonia Nitrogen ($NH_4$-N)

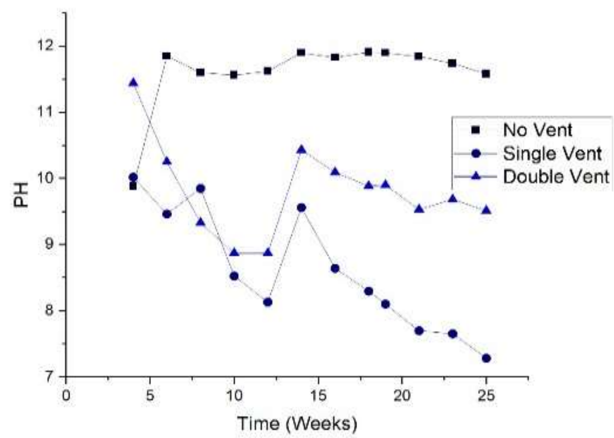



Figure 3-13. Leachate pH

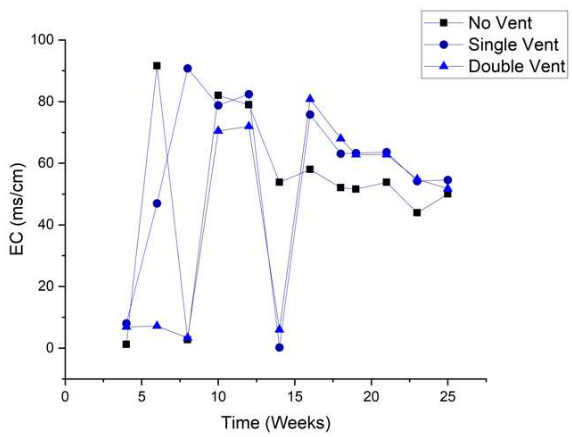



Figure 3-14. Leachate Electrical Conductivity (EC)

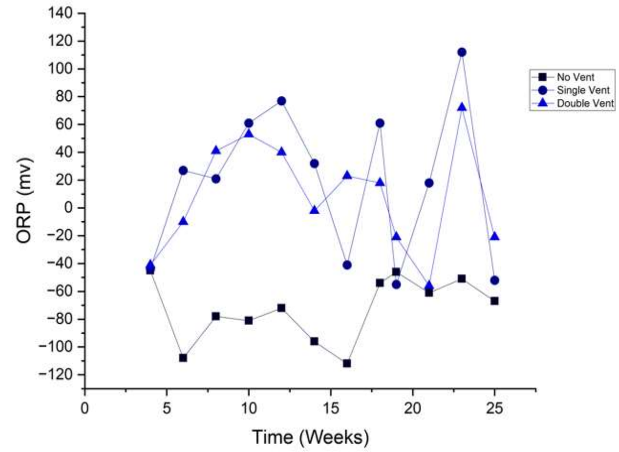



Figure 3-15. Leachate Oxidation-Reduction Potential (ORP)

lysimeter B seemingly demonstrating better absolute TOC reduction. It is surmised that this difference may be attributed to lysimeter B offering better conditions for the proliferation of myriad of heterotrophic bacterial species. The case for the lysimeter B promoting higher microbial growth is further increased when nitrogen is considered as in Figure 3-11& 3-12.

Leachate ammonia nitrogen ($NH_4$-N) is a product of the decomposition of proteins, a process commonly termed ammonification (Kjeldsen et al., 2002). High concentrations are typically associated with mature leachate with an average concentration of 740mg/L (Miao et al., 2019). Initially the lysimeter C generated greater $NH_4$-N but it was surpassed in weeks 8-16 by the single vent lysimeter suggesting that ammonification processes were accelerated by microbial proliferation.

The differences between the Lysimeter B and C could possibly be attributed to coverage provided by the vents. The presence of the second vent in the same volume created an even distribution of oxygen as visualized in Figure 3-8 and quantified by the lower standard deviation in oxygen levels seen in Figure 3-9. The single vent lysimeter, on the other hand, had an uneven oxygen distribution with the unventilated side generally having a lower oxygen concentration. Therefore, an oxygen gradient developed between the ventilated and unventilated areas. The presence of this gradient could then allow for different species of microorganisms whose metabolism is optimized for specific oxygen ranges to develop microbial hotspots. Microbial hotspots can be defined simply as areas where favorable conditions lead to accelerated proliferation of microbes in a specific area compared to the surrounding areas. They have been identified in soil science (Kuzyakov & Blagodatskaya, 2015) and have been specifically researched and verified for their effect on nitrogen cycling (Bonkowski et al., 2000; Groffman et al., 2009; Schlüter et al., 2019).

$NH_4$-N is removed from leachate through nitrification processes which occur in the presence of oxygen as it is converted in successive steps into nitrites then to nitrates. (Ritzkowski et al., 2006) suggested that, under aerated conditions, simultaneous nitrification and denitrification processes lead to significant ammonia nitrogen reduction. Studies on intermittent aeration (IA) show that aerobic and anaerobic conditions do not need to be spatially separated but can be created temporally in the same waste area by alternating the aeration regime (Nag et al., 2016, 2018). In studies by Nag et al., (2016) it was observed that with aeration regimes of 3 days aeration and 4 days without aeration the greatest change in oxygen concentration observed was 20% immediately after aeration to 10% after the 4th day without aeration. Where aerated and anoxic zones occur in the same soil, nitrification-coupled denitrification can take place (Wrage et al., 2001; Zhu et al., 2015) in oxygen atmospheres between 5-15% (Hwang & Hanaki, 2000).

In the present study, lysimeter A was typically in the lower oxygen range seen by Nag et al., (2016), lysimeter C in the higher ranges while lysimeter B spanned most of the range based on the box plot in Figure 3-9. Given the oxygen gradient of the single vent lysimeter it could be concluded that the likelihood of nitrification-coupled denitrification and denitrification hotspots forming was increased. Increased nitrification could then contribute of the lower pH levels for the single vent lysimeter as seen Figure 3-13 as nitrites and nitrates are acidic. While nitrifying bacteria are autotrophic, denitrifying bacteria are heterotrophic utilizing carbon substrates thus increasing TOC reduction.

### 3.3.4 MICROFAUNA

At the end of the experimental period, leachate samples were taken three times at 2–3-week intervals for bacterial counts utilizing a standard agar method. For each sample the general population size of both the aerobic and anerobic species were taken without focus on any genus or species. The result of this analysis is presented below in Figure 3-16 and 3-17. Here it is observed that in all lysimeters both aerobic and anerobic species are coexisting simultaneously. With there being oxygen available aerobic species are dominant. While it would be expected that there would be a direct correlation between oxygen concentration and aerobic microorganism counts this was not obvious in this experiment. In Figure 3-16 while it was true that the ventless lysimeter had reduced levels of aerobic bacteria compared to lysimeters containing vents, the lysimeter with a single vent consistently had the larger population of aerobic bacteria. The difference ranged from 2x to as much as 5x. In Figure 3-17 is observed that anaerobic species are also present in the population despite the presence of aerobic species and oxygen. Anaerobic organisms are classed as either obligate anaerobes where oxygen is toxic, and they cannot exist in its presence, or they are facultative which can exist in presence of oxygen. This suggests that the anaerobic species present were predominantly facultative.

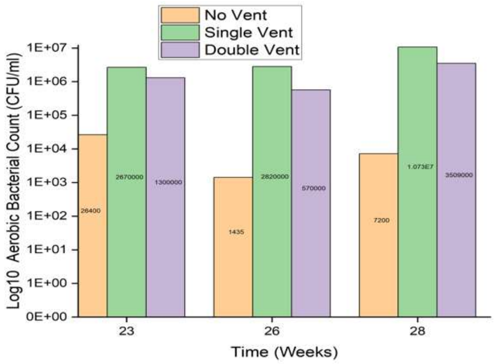



Figure 3-16 Leachate Aerobic bacterial counts

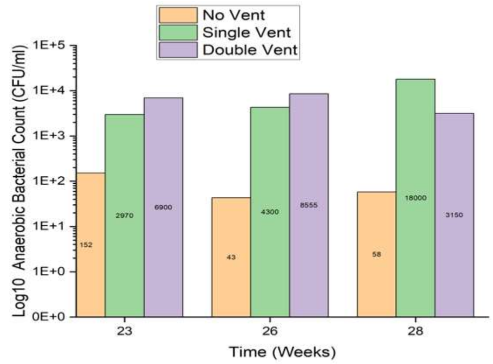



Figure 3-17- Leachate Anaerobic bacterial counts

Denitrification is generally restricted to anoxic environments as oxygen is an inhibitor of the denitrification process. (Hwang & Hanaki, 2000) showed that while presence of oxygen in denitrification reduces the overall denitrification process, it enhances the release of $N_2O$. Also, they showed that biological induced denitrification can occur in atmospheres containing oxygen up to 15%.

If the averaged oxygen concentrations at shown in Figure 3-9 are considered it can be stated that all the oxygen concentrations were within the critical limit for nitrification- denitrification. The main difference would be that in the unventilated lysimeter there was lower concentrations of $NH_4$ available for conversion to nitrates due to reduced aerobic decomposition of waste. Additionally, (Sang et al., 2009) showed that the presence of excess oxygen can produce an inhibitory effect reducing the activity of microbes while simultaneously increasing their utilization of energy.

When (Sang et al., 2009) analyzed the microorganism population of a reactor which utilized intermittent aeration it showed that unique and diverse eubacterial and archaeal communities were formed under intermittent aeration. Experiments undertaken by (Nag et al., 2016) revealed that in intermittent aeration reactors the oxygen concentrations can remain as high as 10% after 4 days of no aeration in high organic waste. As such it can be expected that in intermittent aeration $O_2$ levels are, on average, always above 0 and possibly at or above 10%. The condition in the lysimeters presented in this experiment thus mirror the conditions of intermittent aeration where the $O_2$ levels are maintained above 10% while maintaining zones near 15% near ventilation pipes. The greater the number of zones in the 10% $O_2$ concentration range would result in more diverse microorganism communities as was observed by (Sang et al., 2009)

(Zhang et al., 2023) also noted that paired nitrification-denitrification at oxic-anoxic boarder zones can account for significant proportions of denitrification in organic soils. It has also been postulated that some nitrifiers could, in addition to their role in nitrification, also denitrify as well (Ritchie & Nicholas, 1972). This has been termed as nitrifier denitrification and differs from nitrification-denitrification as pure denitrifying bacteria are not involved. Also, NO-3 is not an intermediate product of nitrifier denitrification. (Wrage et al., 2001)

### 3.4 CONCLUSION

The inclusion of purpose-built ventilation shafts into waste filled lysimeters were effective in delivering oxygen into the lower regions of the waste mass without the need of forced aeration mechanisms.

Increased oxygen in waste generated faster TOC reduction but, as other works on intermittent aeration have shown, could have also inhibited beneficial bacterial processes which require anoxic conditions to proliferate. The creation of both aerobic and anaerobic conditions separated spatially appeared to be an optimal solution. The insertion of passive aeration vents could serve as an alternative to intermittent aeration by removing the need for energy expenditure while still creating similar conditions. The choice in frequency / arrangement of passive vents may affect the balance between adequately aerated and anoxic zones and must be taken into close consideration.

Further work is needed to analyze the extent of denitrification which these passive aeration systems can facilitate compared to traditional intermittent aeration methods.

This investigation showed that the introduction of stand alone vertical gas vents promotes improved aeration of waste masses. Vents can effect up to 1 meter away from the air openings with decreasing effectiveness beyond this range. However a single vent was able to produce greate improvements to leachachate quality and microorganisms proliferation suggesting that denitrification would be optimized in environments as observed in Lysimeter B where oxic-anoxic conditions co-exist, and utilization of passive venting mechanisms can create these optimal conditions with fewer vents.

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# CHAPTER 4

# LYSIMETER AND LANDFILL SIMULATION

# CHAPTER 4

## 4. Lysimeter and landfill simulation

4.1 INTRODUCTION

4.2 MODEL SETUP

4.3 RESULTS AND DISCUSSION

4.4 CONCLUSIONS

## 4.1 Introduction

Landfills are complex systems. This is due firstly to the range of different materials which may have been included in the waste such as easily degradable substances (kitchen waste, green waste, paper) and non-degradable waste. In addition to the variability of the materials which comprise landfills the proportions and distribution of each material within landfills vary significantly within each individual landfill and between landfills. After deposition these materials undergo degradation processes which are affected by local factors such as temperature and rainfall (climate), biota and landfill operating practices. There are various processes operating in landfills which develop and evolve over an extended time span, typically in the range of 1-2 decades. These points make studies and data on the long term processes involved in MSW scarce with knowledge gaps prevalent (Bozkurt et al., 2000b). Thus models are of significant importance in the advancement MSW treatment.

Composting models have been presented in the literature since 1976 (Mason, 2006). Models developed for landfills tend to focus on specific facets of landfill such as gas generation or leachate production. These models can be further specialized into those which focus on aerobic or, more commonly, the anerobic interactions which generate gaseous and liquid products. The focus on anerobic processes was appropriate as anerobic type landfills accounted for most of the landfills which were in operation.

In recent years there has been an increase in the utilization of aerobic techniques in landfill management including both passive and forced aeration technologies(Ritzkowski & Stegmann, 2012). In the absence of these techniques, it was acceptable to utilize anaerobic models as the aerobic phase of landfills was ephemeral, only dominant in the early stage of the life cycle for as long as the initial oxygen present in the landfill remained. However, with the introduction of aerobic and semi-aerobic techniques the aerobic phase can be potentially extended indefinitely(Ritzkowski & Stegmann, 2018)

It is thus necessary to implement landfill models which are capable of emulating environments where aerobic and anaerobic processes are occurring simultaneously. This work attempted to implement a landfill simulation where aerobic and anaerobic processes were simulated simultaneously by combining existing preexisting models.

## 4.2 Simulation construction

### 4.2.1 Model selection criterion

An important consideration in the selection of the model utilized is the type of waste which is expected to be dominant in the waste mass. Waste low in easily degradable matter is typically dominant in holocellulose which consists mainly of the complex carbohydrates, cellulose and hemicellulose from paper and wood.(Barlaz, 2006). This type of waste is characteristic of developed countries. Developing countries differ where kitchen waste predominates. In addition to the holocellulose, kitchen waste consists also of high proportions of easily degradable non- holocellulose compounds including simple sugars, lipids and amino acids. Compost is analogous to kitchen waste and (Mason, 2006) performed a comprehensive review of models that have been traditionally applied to the composting process. The trend common to most models is the use of Monod kinetics to incorporate the importance of relationship between substrates and microbial population proliferation.

Another important selection criteria was the reaction processes expected to be prevalent in the waste mass. Many MSW degradation models focus on the anaerobic respiration processes in waste. This was logical due to most landfills in operation being anaerobic landfills. Much of the life cycle of anaerobic landfills occur in the absence of oxygen. Aerobic processes occur in the earlier phase of the life cycle of but are short lived. In the previous chapter of this study, it was validated that the presence of the ventilation pipes can improve delivery of air to the portion of the waste mass in contact with the pipes. This makes aerobic models appropriate. However, the effective radius of this is limited thus anaerobic or low oxygen zones are expected to develop in intermediate regions between pipes and areas below the vertical extents of the pipes. Thus, there is also a need for anaerobic models to incorporate zones of low oxygen concentrations. Therefore, this study incorporated models which consider both reaction models simultaneously. Single type models were presented by (Pommier et al., 2007, 2008a) however dual reaction process models presented by (Kim et al., 2007) were found to be most applicable and widely referenced in other studies.

#### 4.2.2 Simulation structure

(Kim et al., 2007) presented a one-dimensional compartment model. The advantage of these models can lie in that a smaller unit of the landfill (cell) can be simulated to represent the heterogeneity of the entire landfill. This model It integrates processes such as organic biodegradation, bacterial activity, solute transport as shown in Figure 4-1. In the model the landfill was separated into 3 phases: Solid, liquid and gas phases. In the solid phase the deposited waste was first considered as undegraded organic matter which is hydrolyzed into easily degradable form. The chemical formula of the waste was represented by the simplified chemical formula $C_6H_9ON$ which has been adapted in other studies. (Fytanidis and Voudrias, 2014). Also considered in the solid phase were microbial biomass. In this simulation five types of biomass were considered based on their preferred substrate and affinity for oxygen. the simplified chemical formula $C_5H_7NO_2$ was assigned to all biomass.(Fytanidis and Voudrias, 2014; Stephanopoulos, G., Aristidou, A.A., & Nielsen, 1998). Dissolved organic carbon is firstly and decomposed into $CO_2$ gas by aerobic bacteria in the liquid phase. This aerobic decomposition releases ammonia into the liquid phase which is subsequently transformed into nitrite by nitrous bacteria, and nitrite into nitrate by nitrifying bacteria under aerobic conditions. Under anoxic conditions nitrate is transformed into nitrogen gas by denitrifying bacteria. Under anaerobic conditions methane and carbon dioxide are generated by anaerobic bacteria in the liquid phase. Gas transport inside the lysimeter is dominated by diffusion and convection.

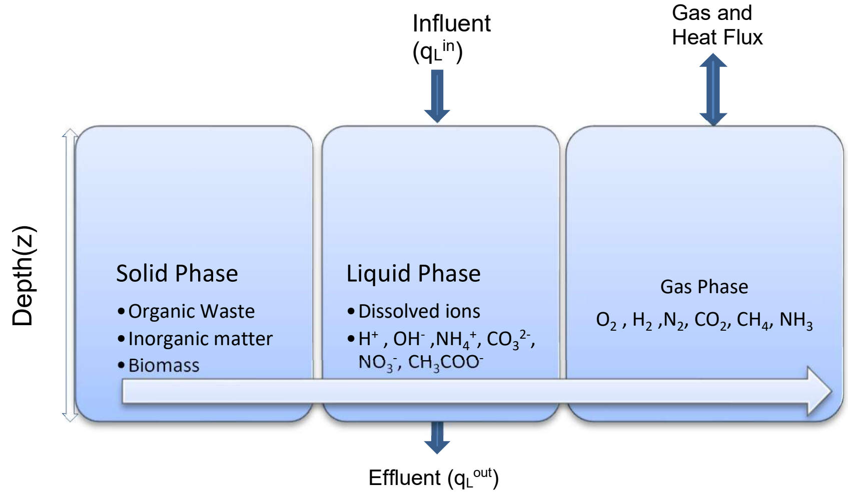



Figure 4-1. Landfill Model Outline

### 4.2.3 Kinetic models

Hydrolysis

In the case of particulate matter, hydrolysis is normally considered as the rate-limiting step (Pommier et al., 2008b). Hydrolysis may consist of many separate processes and traditionally they are collated in a simple first order reaction for substrate degradation. However, it was observed in tests with high levels of inoculum that hydrolysis rates become elevated showing biomass concentration has a degree of control on hydrolysis. (Vavilin et al., 2008). A model was proposed by (Lin et al., 2008) which factors the influence of temperature on biomass involved in hydrolysis:

$$R_{\mathrm{h}} = K_{\mathrm{h}} * S_i * \mathrm{h}tf \qquad \text{(Eq.4 -1)}$$

where $R_{\mathrm{h}}$ is the rate of hydrolysis at T °C (mol m-3 s-1), $K_{\mathrm{h}}$ the hydrolysis rate constant (s-1), $S_i$ in insoluble substrate concentration (mol m-3 ), and $\mathrm{h}tf$ the temperature function utilized to express the effect of temperature on the rate of hydrolysis (-). The temperature function utilized was expressed as

$$\mathrm{h}tf = 0.0182*T \quad (5\ ^{\circ}C < T < 75\ ^{\circ}C) \qquad \text{(Eq.4 -2)}$$

where hydrolysis bacteria have an optimum range between 5-75 °C

Growth and Oxygen consumption

Monod kinetics are normally used to describe substrate limiting growth of microbial populations in landfills as follows:

Aerobic Decomposition $\mu\_max \frac{S}{Ks+S} \frac{O_2}{Ko+O_2}$ X_I gtf (Eq.4 -3)

$NH_3$ to $NO_2$ $\mu\_AOB_1 \frac{O_2}{KO_2\_AOB+O_2} \frac{NH_3}{KNH_3\_AOB+NH_3}$ X_AOB gtf (Eq.4 -4)

$NO_2$ to $NO_3$ $\mu\_NOB \frac{O_2}{KO_2\_NOB+O_2} \frac{NO_2}{KNO_2\_NOB+NO_2} *$ X_NOB gtf (Eq.4 -5)

$NO_3$ to $NO_2$ $\mu\_max\ n\_HB1 \frac{KO_2\_i}{KO_2\_i+O_2} \frac{S}{Ks\_1+S} \frac{NO_3}{KNO_3\_HB+NO_3}$ X_ANOX gtf (Eq.4-6)

$NO_2$ to NO $\mu\_max\ n\_HB2 \frac{KO_2\_i}{KO_2\_i+O_2} \frac{S}{Ks\_2+S} \frac{NO_2}{KNO_2\_HB+NO_2}$ X_ANOX gtf (Eq.4-7)

NO to $N_2O$ $\mu\_max\ n\_HB3 \frac{KO_2\_i}{KO_2\_i+O_2} \frac{S}{Ks\_3+S} \frac{NO\_}{KNO\_HB+NO}$ X_ANOX gtf (Eq.4-8)

$N_2O$ to $N_2$ $\mu\_max\ n\_HB4 \frac{KO_2\_i}{KO_2\_i+O_2} \frac{S}{Ks\_4+S} \frac{N_2O}{KNO\_HB+N_2O}$ X_ANOX gtf (Eq.4-9)

*Acetogenesis* $\mu\_an \frac{KO_2\_i}{KO_2\_i1+O_2} \frac{S}{Kan+S}$ X_AN gtf (Eq.4 -10)

Methanogenesis $\mu\_me \frac{KO_2\_i}{KO_2\_i2+O_2} \frac{CH3COOH}{Kme+CH3COOH}$ X_ME gtf (Eq.4 -11)

Where μ_(*) (1/h) is the maximum specific growth rate of different bacteria, S(kg–C/$m^3$) the concentration of dissolved organic carbon. K(*) (kg–substrate/$m^3$) is the half-saturated coefficient of different substrates. (* = C, $O_2$, $NH_3$, NO, $NO_2$, $NO_3$ and $O_2$) for the respective bacteria involved in that process. $KO_2\_i$ (kg–$O_2$/$m^3$) is the Oxygen inhibition coefficient for anoxic processes where oxygen is inhibitory.

X_(*) (kg–cell/m3) is the concentration of bacteria involved in each respective process. (* = Aerobic, Ammonia Oxidizing, Nitrate Oxidizing, denitrifying, Acetogenic and Methanogenic) and gtf(-) the generalized growth temperature function for bacteria.

The growth temperature function is given by

$$\frac{(T-T_{max})*(T-T_{min})^2}{(T_{opt}-T_{min})*[(T_{opt}-T_{min})*(T-T_{opt})-(T_{opt}-T_{max})*(T_{opt}+T_{min}-2T)]} \quad \text{(Eq.4 -12)}$$

Were $T_{opt}$ is the optimal growth temperature, $T_{min}$ the minimum growth temperature and $T_{max}$ the maximum growth temperature. These values are generally accepted to be 58.6 , 5 and 71.6 °C respectively.

### 4.2.4 Biomass model

The main factor which affects the substrate concentration in waste is the biomass concentration. Microorganisms utilize the organic components of waste as raw materials for their growth. With readily available substrates the biomass concentration increases at a rate determined by the Monod reaction. Being living organisms and comprised of organic matter biomass dies and is subject to degradation and decay. The rate of biomass creation is therefore balanced by biomass decay to ensure biomass concentrations were dynamic, as in nature, and the removing the possibility of negative growth rates the biomass decay model proposed by (S. Y. Kim et al., 2007) was implemented. This model consists of two main equations, biomass growth and biomass decay. The change in biomass concentration was determined by their difference.

Biomass growth $R_g$ $\quad R_n * y_n$ (Eq.4 -13)

Biomass decay $R_d$ $\quad bd_n * X\_n \quad * \text{dtf}$ (Eq.4 -14)

Where $R_n$ (kg/m$^3$*s) is the growth rate of biomass as determined from the Monod kinetic reactions (Equations 4-4 to 4-12) , y (-) the Yield coefficient for microorganism *(n)* based on their substrate, bd(1/d) the biomass decay constant for microorganism *(n)* , X (kg–cell/m3) the concentration *(n)* of bacteria and dtf(-) the generalized decay temperature function for bacteria. The decay temperature function is given by

$2.14 \times 10^{-4}T^2 - 2.356 \times 10^{-2}T + 1.348$ (5 °C< T < 75 °C) (Eq.4 -15)

Biomass growth was determined by the Monod equation where decay is optimal in the same range as growth, between 5-75 °C

### 4.2.5 Heat transfer

The degradation of waste is an exothermic process which generates heat. This heat is released to the waste mass and drives gas convection. Heat transfer in porous media was included in the simulation where

$$\rho c_p \frac{\partial T}{\partial t} = \nabla \cdot (k \nabla T) - \rho c_p \mu \cdot \nabla T + \dot{q}_m \qquad \text{(Eq.4 -16)}$$

Where ρ(kg/m$^3$) is the density of the porous medium. $c_p$(J/kg*K) the specific heat, T(K) the temperature T(s) the time, k(W/m*K) the thermal conductivity of the medium, u(m/s) the velocity of fluid flow in medium, $\nabla T$ (K/m) as the temperature gradient and $\dot{q}_m$(W/m$^3$) representing any other heat sources or sinks present.

### 4.2.6 Transport model

For predicting gas pressure, it is assumed that gas transport would obey Darcy's law. In the simulation package a modification of the Navier-Stokes equation known as the brinkman equation was utilized. This was due to the presence of the gas vents within the simulation. Darcy's equation are suited for porous medium but the hollow pipes are not suited for this equation only hence the utilization of the Brinkman equation which incorporates both fluid flow and porous flow. The Brinkman equation is given by:

$$\rho(\mu \cdot \nabla)u = -\nabla p + \mu\nabla^2 u - \rho\alpha\mu + pg \qquad \text{(Eq.4 -17)}$$

Where ρ(kg/m$^3$) is the density, (μ · ∇)u (m/s$^2$) term refers to acceleration due to convection, $\nabla p$ (Pa/m) accounts for the pressure gradient $\alpha$(-) refers to permeability and $\mu$ the dynamic viscosity (kg/m*s) and $g$ the effect of gravity(N/m$^3$

Diffusion was expected to follow Fick's law of diffusion. The species in the model are involved in reactions while being transported so this was taken into consideration utilizing the species conservation equation which is typically used to consider mass transfer and reactive transport. This was represented by

$$\frac{\partial}{\partial t}(\rho\omega_i) + \nabla \cdot (p\omega_i u) = -\nabla \cdot j_i + R_i \qquad \text{(Eq.4 -18)}$$

Where $\frac{\partial}{\partial t}(\rho\omega_i)$ (kg/m$^3$*s) represents the rate of change of mass fraction with respect to time with $\omega_i$ (-) the mass fraction of species $i$ and ρ(kg/m$^3$) the density. The $\nabla \cdot (p\omega_i u)$ (kg/m$^3$*s) term represents convective flux which is a product of pressure p(Pa), mass fraction and velocity $u$(m/s). $-\nabla \cdot j_i$ (kg/m$^3$*s) is the diffusive flux of species $i$. $R_i$(kg/m$^3$*s) is a reaction term for any additions or removals of species $i$ due to other chemical reactions and processes.

### 4.2.7 Simulation Application

All modelling was undertaken using the modelling software COMSOL Multiphysics® version 5.4. utilizing the following modules to incorporate the models outlined previously.

- Computational Fluid Dynamics (CFD)
- Chemical Reaction Engineering
- Subsurface Flow
- Heat Transfer
- Porous Media Flow

#### 4.2.8 Limitations

For simplification purposes some aspects which may be presented in other models where not actively considered in the model created here. Some aspects which may be considered as limitations include.

a. the effect of pH and its inhibition of some reactions was not considered.

b. while biomass growth was considered the individual stoichiometric processes of biomass growth were not considered. Where in biomass growth ammonia released from solid waste would become incorporated into biomass these reactions were not modeled and as such it could be expected that modeled results overestimate the concentrations of some substances that would have been incorporated into biomass.

### 4.3 Results and Discussion

#### 4.3.1 Lysimeter Experiment Simulation

To determine the accuracy of the created model, it was utilized to recreate the 2D lysimeter experiment detailed in Chapter 3. The simulation focused on the $O_2$ characteristics of the lysimeter as $O_2$ concentrations play a critical factor in both nitrification-denitrification as well as anaerobic processes.

For simplification of the model the following assumptions were utilized.

a. Heat :- lysimeter perimeter = No Flux ; top of lysimeter and vents = open boundary

b. Pressure :- lysimeter perimeter = No Flux ; top of lysimeter and vents = open boundary

c. $O_2$ mass fraction = 0.21 at all zones at t = 0

d. $N_2O$ mass fraction = 0 at all zones at t = 0

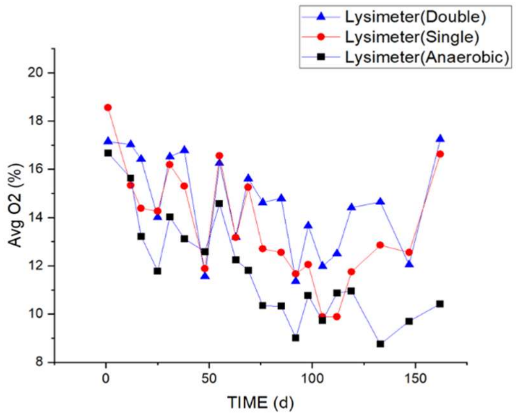
e. Temperature =273K at t=0

f. Pressure = 1 atm at t=0

g. Water content was constant and was not a limiting factor

Other parameters utilized in the simulation are presented in the appendix.

Due to the importance of oxygen in both the promotion of nitrification and the inhibition of denitrification it was assumed that recreating similar oxygen profiles as seen in the lysimeter experiment would enable related processes to be modelled affectively. Figure 4-2 and 4-3 show the comparison of the mean oxygen levels measured from the lysimeter experiment against the mean oxygen levels calculated via the simulation for each lysimeter respectively. Like the experimental data the simulated quantity was averaged only across the waste layer. Simulated values captured the overall trend with mean $O_2$ increasing with the number of vents showing a fit with experimental data.



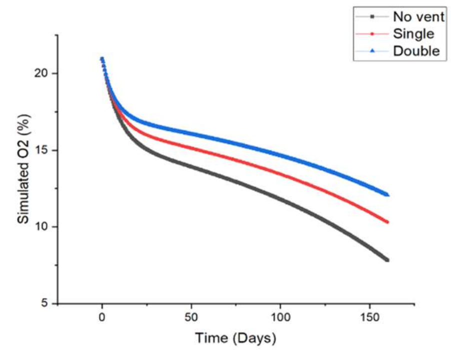




Figure 4-2. Mean measured $O_2$ concentration Figure 4-3. Mean $O_2$ simulated.

(Mason, 2006) found that current models are unable to properly model oxygen and carbon dioxide profiles and that more work is needed to incorporate the effects of natural aeration into composting models. This can thus explain the smoothness of the simulated results. However, despite this the simulations show a good fit with the experimental data and can be utilized for further investigations.

### 4.3.2 LANDFILL SIMULATION

With the COMSOL generated model being validated utilizing the data obtained from the lysimeter experiment it can then be assumed that the model can be extrapolated to attempt to model a larger landfill mass. While Chapter 3 showed that the ventilation pipes can be effective on a small-scale present in the lysimeters the magnified scale present in a landfill may result in performance being affected. In addition, creation of this model can allow for additional benefits such as

- Determining optimal pipe depth to landfill depth
- Determining optimal pipe diameter
- Determining optimal pipe spacing.

Having a model which can be the basis for determining the above parameters would prove valuable if passive venting is implemented on a practical scale as it would allow for significant time and resource savings as the desired configuration can be tested and validated by the model before implementation.

For all simulations a theoretical 2D landfill of size 50M wide by 20M deep was utilized. This size was derived in part from considering some landfill sizes described in literature but was mainly utilized based on modelling trials were where performed to attempt to find a model size to minimize the computation time. Also, to reduce simulation time initial simulations were kept at 180 days initially to for the groups of simulations and extended to compare the best identified scenarios. Simulations were taken in seven-day steps.

### 4.3.3 Pipe depth

The outcome desired from this simulation was to determine what percentage of the total landfill vertical depth should be penetrated by ventilation shafts to ensure maximal coverage while keeping the piping length minimal if possible. Landfills and uncontrolled dumping sites in developing countries would have variable depths based on the length of time they have been in operation. For extremely deep landfills it may be impractical to drill to the base of these landfills. As such it is important to optimize the depth by utilizing the minimum most viable distance.

For this simulation five (5) scenarios were considered.

- 50% of landfill depth
- 60% of landfill depth
- 70% of landfill depth
- 80% of landfill depth
- 95% of landfill depth

Figure 4-4 to 4-8 shows the representation of the five pipe length simulations. For these simulations pipe diameter was kept standardized at 0.5m and pipes were spaced evenly at 15m apart. For each simulation the results for total oxygen, nitrogen, methane and ammonia and were compared to a baseline simulation where no pipes were present.

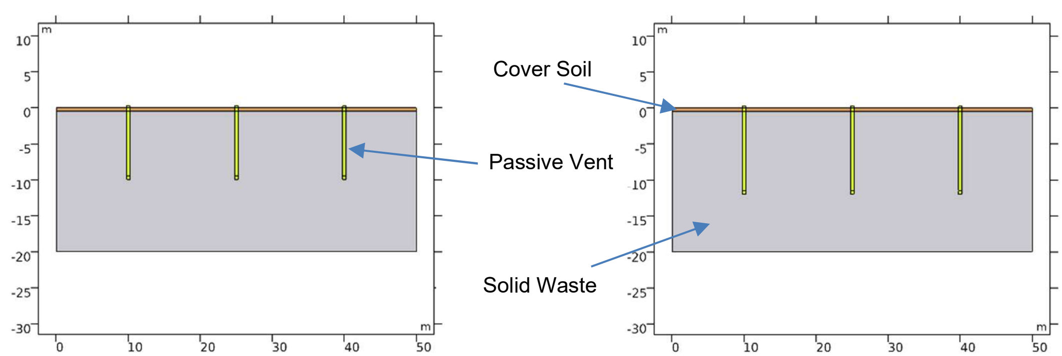



Figure 4-4. 50% depth scenario

Figure 4-5 ..60% depth scenario

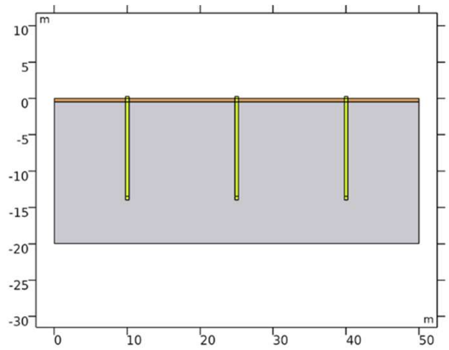



Figure 4-6. 70% depth scenario

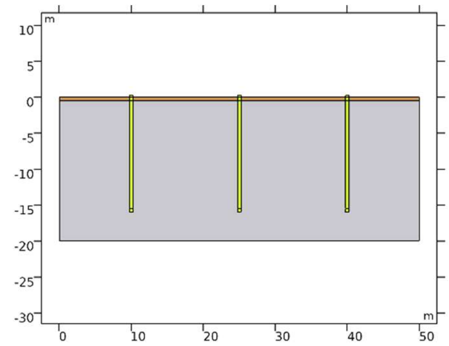



Figure 4-7. 80% depth scenario

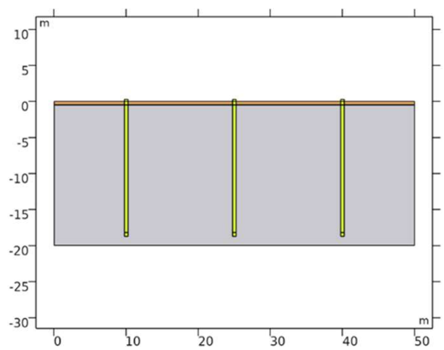



Figure 4-8. 95% depth scenario

Figure 4-9 presents the results of the simulation on the_effect of pipe length on the concentration of the above-mentioned compounds at the end of the simulation period. It was observed that the difference depth of the vents made was muted the most optimal depth based on the simulation was 60% depth. The simulation also showed that the presence of the pipes increased oxygen concentrations by approximately 38%. With the higher oxygen concentrations increased aerobic degradation occurs which releases ammonia. Thus, in Figure 4-10 we see high ammonia levels projected which then results in higher simulated nitrogen. Simulated nitrogen in Figure 4-11 increased by as much as 9.8% compared to the no vent scenario. When methane was simulated, unexpected results were obtained with the unventilated landfill showing the lowest concentration as seen in Figure 4-11. This was thought to be due to the temperature factors utilized for the kinetic reactions. Ventilated landfills should have higher temperatures through the landfill from the heat of aerobic decomposition which when distributed across the cell creates better conditions for anaerobes to operate in low oxygen zones. Also, $CH_4$ production is typical of the late stages of landfills, so the simulation period would not be adequate to fully study methane dynamics.

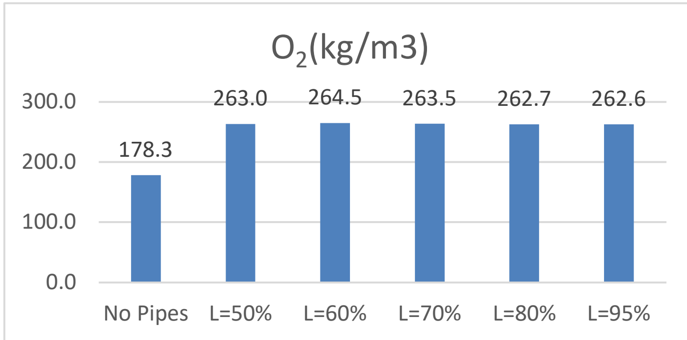



Figure 4-9. Total $O_2$ after 180 days

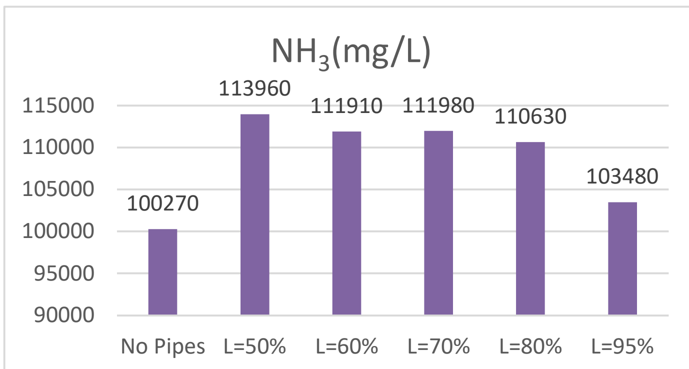



Figure 4-10. Total $NH_3$ after 180 days

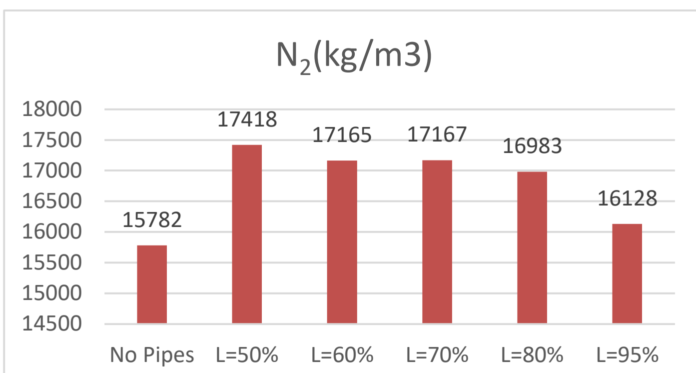



Figure 4-11. Total $N_2$ after 180 days

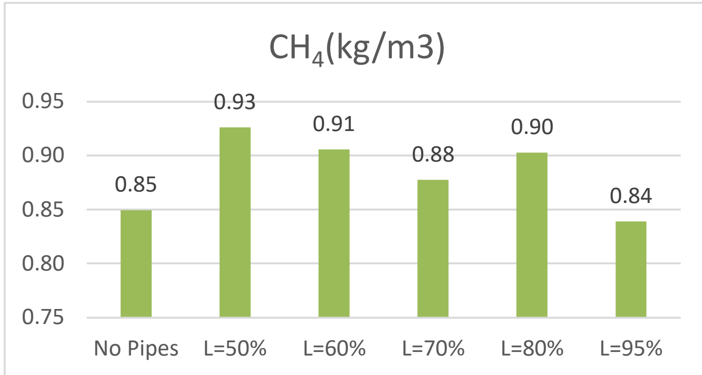



Figure 4-12. Total $CH_4$ after 180 days.

### 4.3.4 Pipe diameter

This simulation was implemented to determine if pipe diameter plays a significant role in delivering air to the deepest regions of a landfill from the surface.

For this simulation four (4) scenarios were considered.

- 0.25m diameter
- 0.5m diameter
- 0.75m diameter
- 1m diameter

Figures 4-13 to 4-16 shows the representation of the four pipe diameter simulations. For these simulations, pipe depth was kept standardized at 95% depth as the objective was to determine which diameter is most effective at depth. Pipes were spaced evenly at 15m apart. For each simulation the results for total oxygen, nitrogen, ammonia and methane were compared to a baseline simulation where no pipes were present.

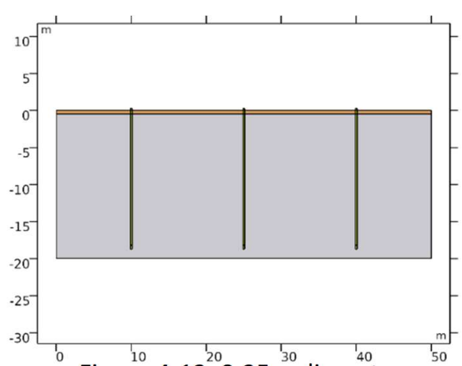


Figure 4-13. 0.25m diameter

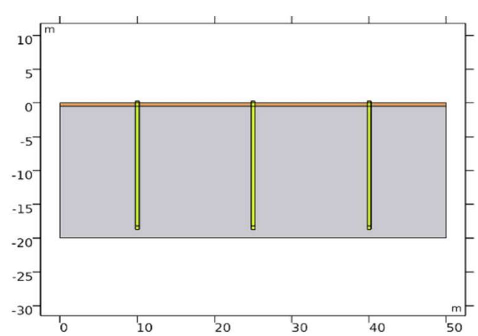


Figure 4-14. 0.5m diameter

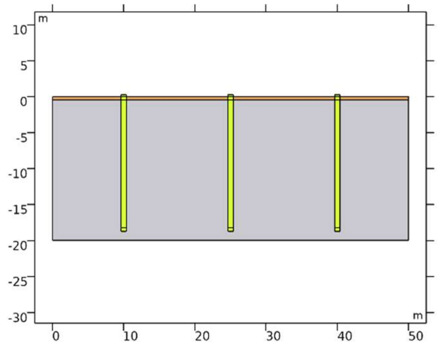


Figure 4-15. 0.75m diameter

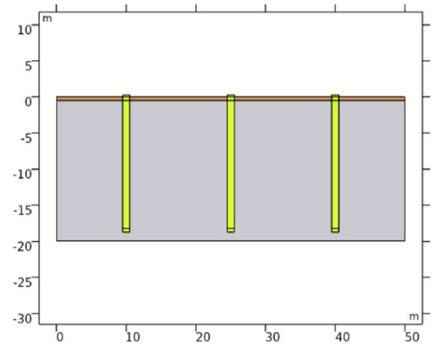


Figure 4-16. 1m diameter

In Figures 4-17 to 4-20 the results of the simulation on the effect of pipe diameter on subsurface oxygen is displayed. It is immediately noticeable that the results from the 0.25m diameter pipes stood out from the rest of the simulations. The concentrations of all compounds were higher at the end of the simulation period. Again, seeing increased methane production even though oxygen levels were higher further reinforces the argument that the temperatures produced from the aerobic decomposition accelerates other processes including anoxic processes. The almost 2-fold increase in mean $O_2$ observed in 0.25m diameter pipe could be attributed to the lower diffusion path length gas needs to travel in thinner pipes. The larger pipes allowed for easier escape of gases from the higher pressure within the landfill which could have contributed to the reduced concentration of other gases as pipe diameter increased.

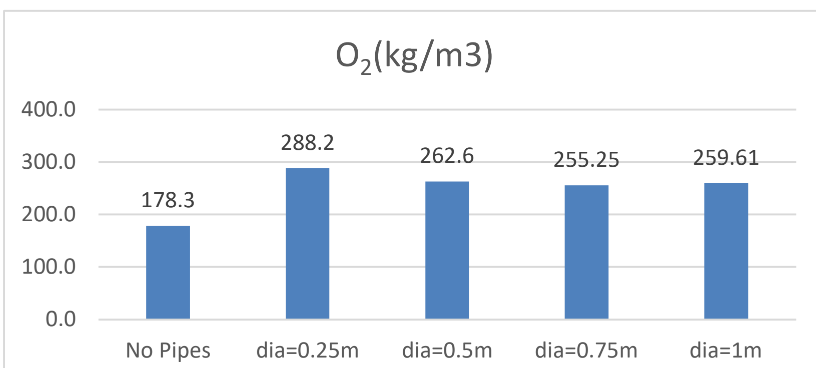



Figure 4-17. Pipe diameter effect on simulated $O_2$ concentration

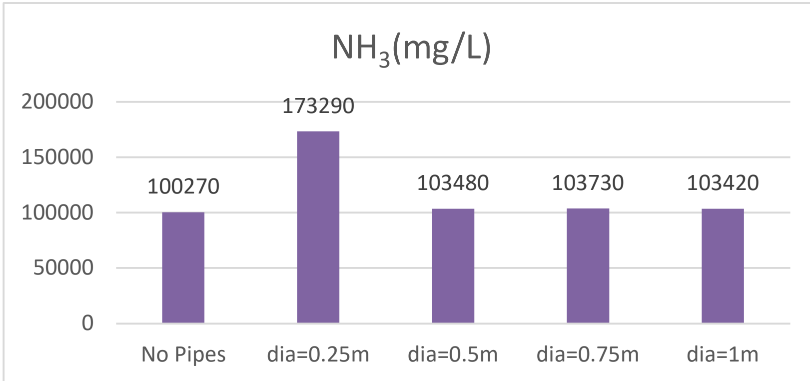



Figure 4-18. Pipe diameter effect on simulated $NH_3$ concentration

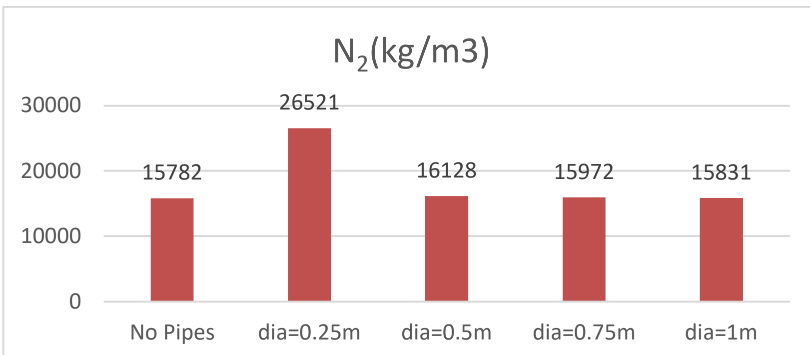



Figure 4-19. Pipe diameter effect on simulated $N_2$ concentration

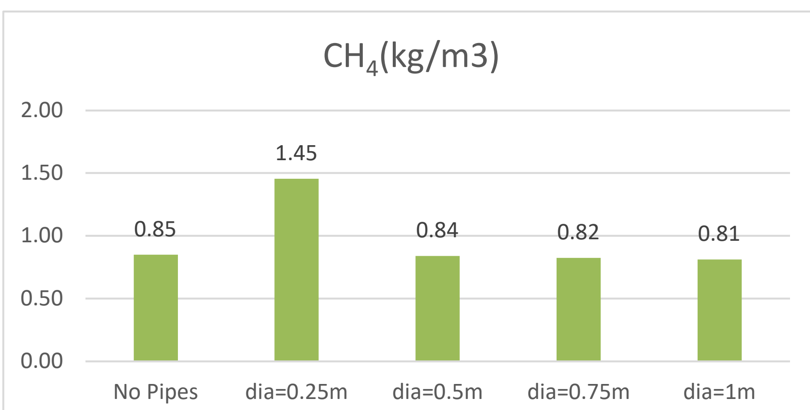



Figure 4-20. Pipe diameter effect on simulated $CH_4$ concentration

### 4.3.5 Pipe spacing

This simulation was implemented to determine the optimal distance apart for passive ventilation pipes. While it may be expected that having as many vents as possible would be ideal this scenario is not thought to be optimal. Firstly, it will increase costs, secondly it will make the landfill difficult to navigate if too many pipes are present and protruding and thirdly as demonstrated earlier in Chapter 3 there is an optimal balance to be found between oxygen concentration and anoxic zones.

For this simulation six (6) scenarios were considered.

- 10m spacing
- 15m spacing
- 20m spacing
- 25m spacing
- 10m spacing with multi-level pipes
- 25m spacing with multi-level pipes

Figures 4-21 to 4-26 shows the representation of the four pipe spacing simulations. For these simulations pipe diameter was kept standardized at 0.5m and depth kept standard at 95%. For each simulation the results for total oxygen, nitrogen, methane and ammonia and were compared to a baseline simulation where no pipes were present.

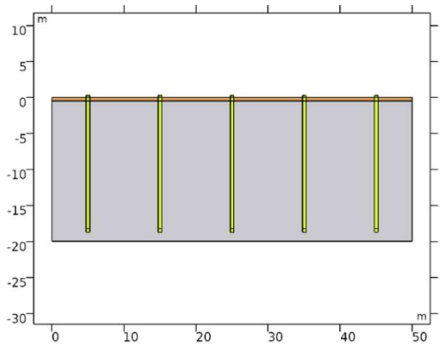



Figure 4-21. 10m spacing.

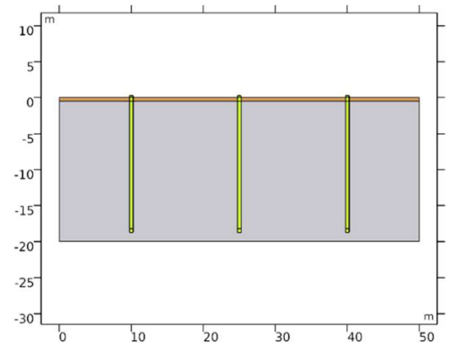



Figure 4-22. 15m spacing.

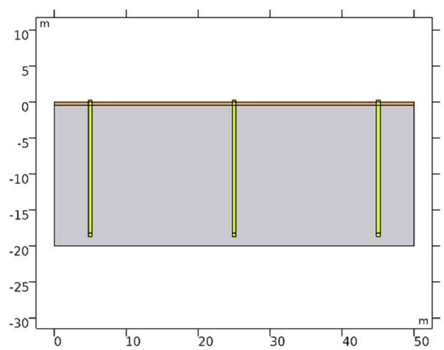



Figure 4-23. 15m spacing.

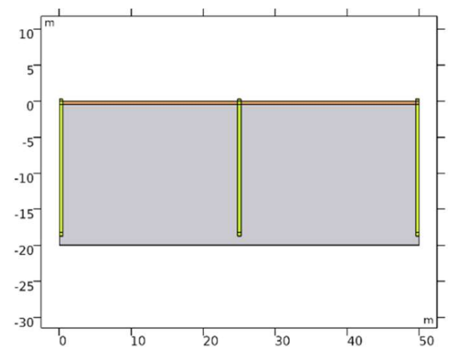



Figure 4-24. 20m spacing.

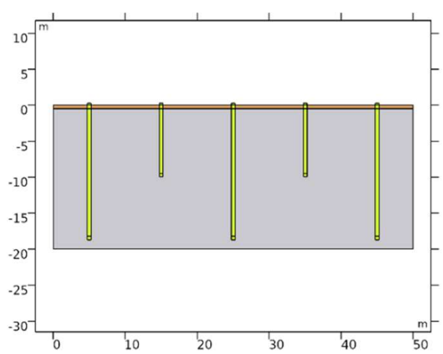



Figure 4-25. multi-level pipes (10m-v2)

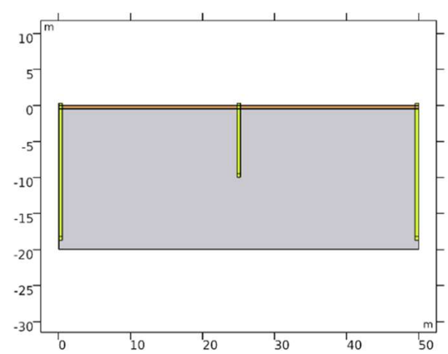



Figure 4-26. multi-level pipes (25m-v2 spacing)

In Figure 4-27 to 4-30 the results of the simulation on the_effect of pipe spacing. The simulations showed that in terms of improving the oxygenation and denitrification having as pipes in a 10m spacing arrangement provided a 53% increase in oxygen concentration compared to the unventilated case. With respect to Nitrogen the difference was 6% even with the pipes also acting as vents for gas release as well. With the modification of the pipe arrangement utilizing alternating long and short vents it was observed that this arrangement was able to introduce a slightly increased level of oxygen compared to the case of only long vents. Higher concentrations of all substances in this configuration also suggests that both anoxic and aerobic conditions were able to coexist in this configuration.

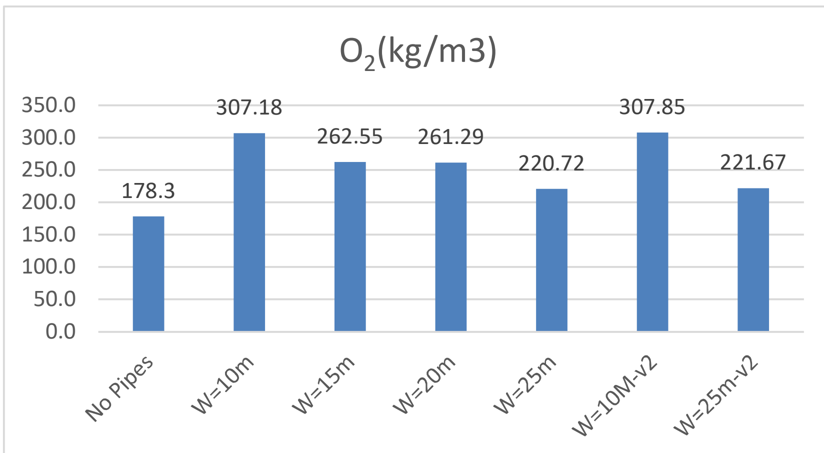



Figure 4-27. Pipe spacing effect on simulated $O_2$ concentration

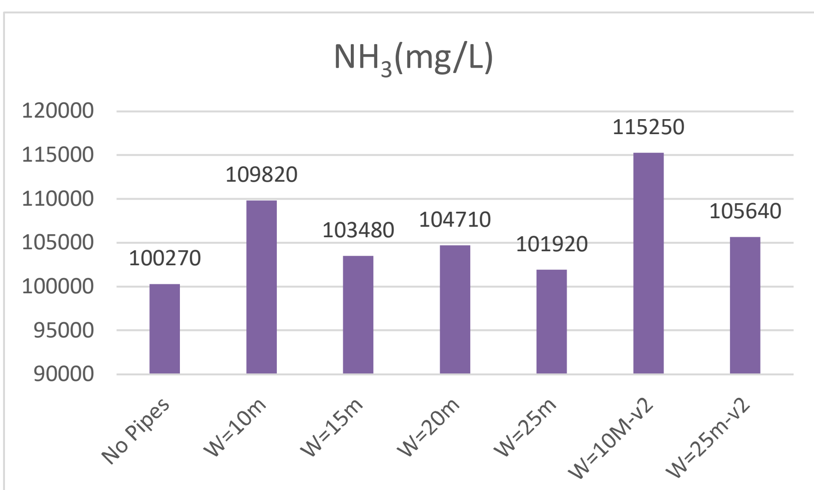



Figure 4-28. Pipe spacing effect on simulated $NH_3$ concentration

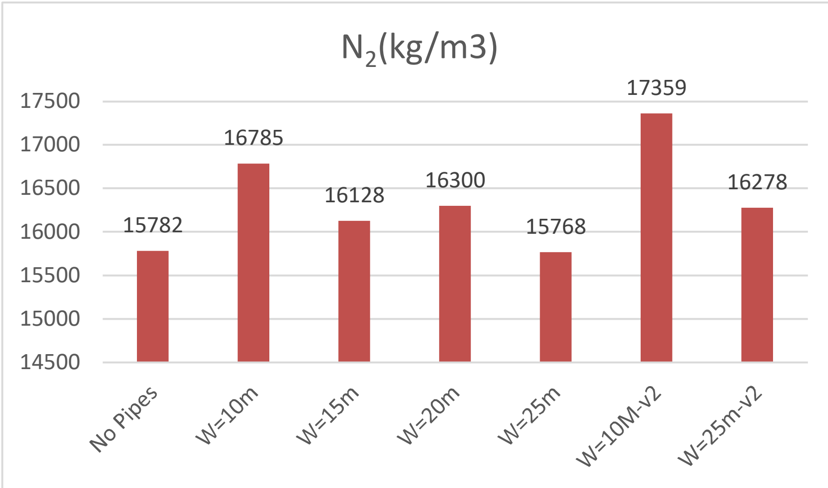



Figure 4-29. Pipe spacing effect on simulated $N_2$ concentration

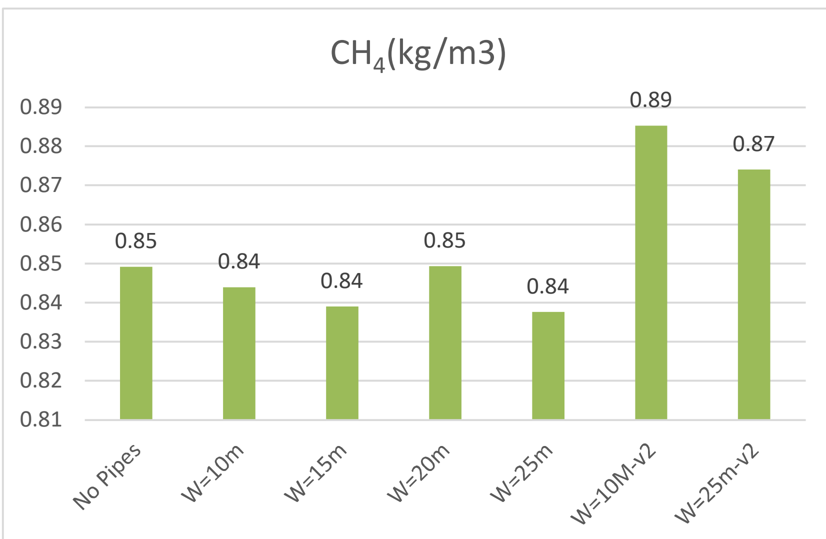



Figure 4-30. Pipe spacing effect on simulated $N_2$ concentration

#### 4.3.6 Proposed Configuration

Based on the simulation data a possible one baseline arrangement for the installation of passive ventilation pipes into landfills was proposed. Figure 4-31 proposes one optimal arrangement for implementation of passive venting in landfills.

Main points of the arrangement are:

- Pipe diameter = 0.25m
- Pipe spacing = 10m spacing
- Pipe arrangement = Alternating deep and shallow pipes.
- Pipe depth = 50% and 80% of landfill depth

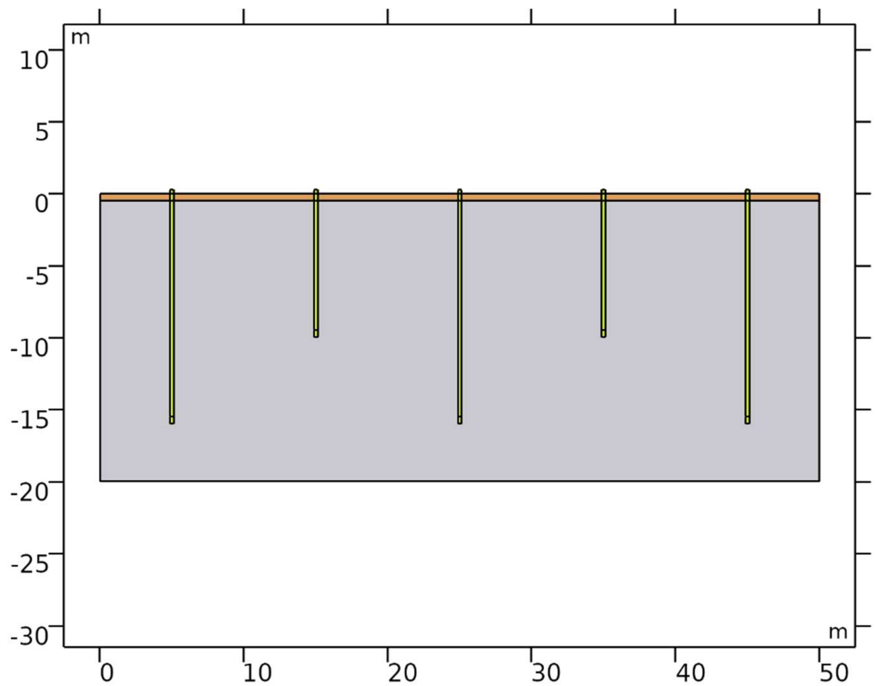



Figure 4-31. Proposed Optimal passive venting arrangement for organic landfills.

A 365-day simulation was then undertaken utilizing this configuration and compared to the case of no ventilation being present. The results of these simulations are presented in Figures 4-32 to 4-36

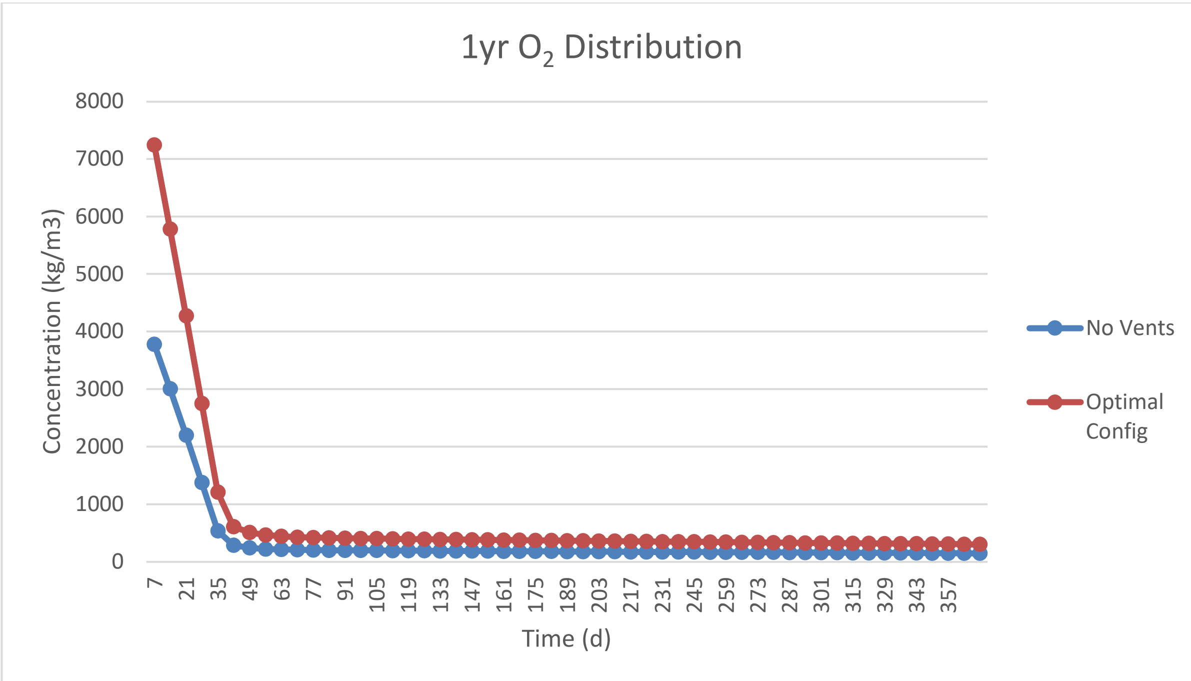



Figure 4-32. Simulated $O_2$ in optimal vs no vent conditions over 1 yr. period

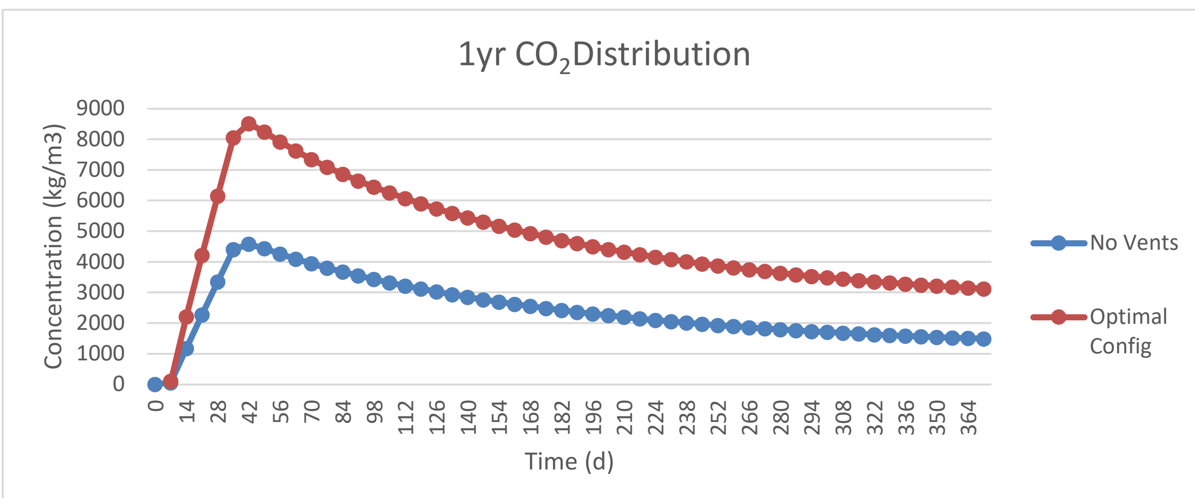



Figure 4-33. Simulated $CO_2$ in optimal vs no vent conditions over 1 yr. period.

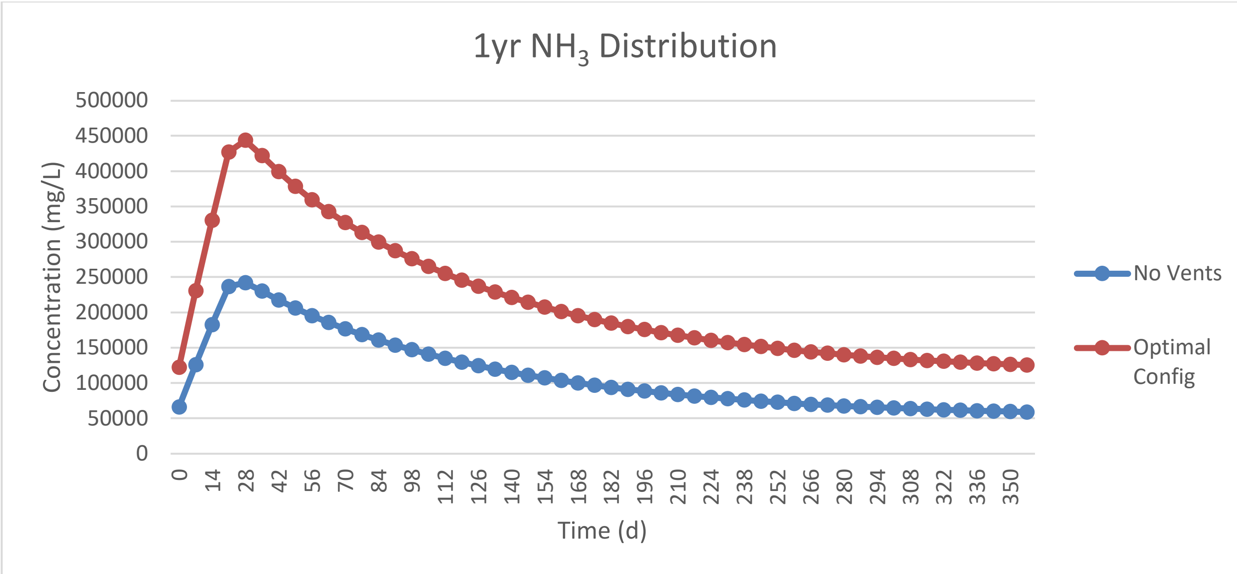



Figure 4-34. Simulated $NH_3$ in optimal vs no vent conditions over 1 yr. period.

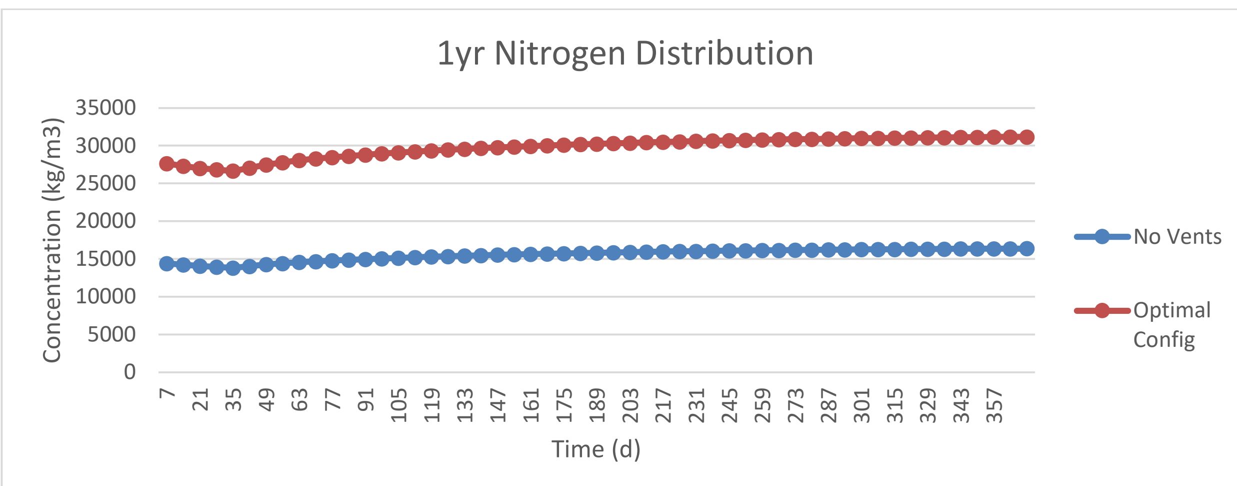



Figure 4-35 Simulated $N_2$ in optimal vs no vent conditions over 1 yr. period.

## 4.4 CONCLUSIONS

Simulations showed that the optimal vent configuration can lead to a consistent increase in oxygen concentrations of approximately 69% over the unventilated condition. While the magnitude of the improvement derived from the simulation may be exaggerated as the simulation was not an exact fit the fact that an improvement was detected is promising.

With increased oxygen concentrations we could expect this to have a chain effect of increased aerobic degradation which produces higher ammonia concentrations. This ammonia is then able to undergo denitrification processes and be converted to nitrogen. This also led to the simulated 63% increase on average in Nitrogen concentrations which may suggest that denitrification was increased by this amount because of the presence of the vents.

On the negative side $CO_2$ production is expected to also be higher which when released to the environment can have a greenhouse effect. However, $CO_2$ is preferred to the long-term methane production which is typical of anaerobic landfills. With increased aerobic decomposition the quantity of waste that will be converted to methane in the long term will be reduced.

Overtime this apparent increase in aeration and acceleration in denitrification could prove significant in reducing the contaminant load from landfills and reducing their aftercare durations.

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## APPENDIX A

### TABLE A1. COMPOUNDS UTILZED IN SIMULATION

| Compound | Formula | Source |
|---|---|---|
| Solid Waste ($S_s$) | $C_6H_9O_{3.56}N_{0.32}$ | (Komilis et al., 2012) |
| Biomass (X) | $C_5H_7O_2N$ | (Sole-Mauri et al., 2007) |

SIMULATED REACTION CHEMISTRY

1. Aerobic bacteria growth
   $S_s+6.23O_2 => 6CO_2+0.32NH_3+4H_2O$

2. $NH_3$ oxidation to $NO_2$:
   $2NH_3+O_2 => NO_2+3H$

3. $NO_2$ oxidation to $NO_3$
   $NO_2+H_2O => NO_3+2H$

4. Anoxic $NO_3$ reduction to $NO_2$
   $NO_3+2H => NO_2+H_2O$

5. Anoxic $NO_2$ reduction to NO
   $NO_2+2H => NO+H_2O$

6. Anoxic NO reduction to N2O
   $2NO+2H => N_2O+H_2O$

7. Anoxic N2O reduction to N2
   N2O+2H=>N2+H2O

8. Acetogenesis
   $S_s+3H_2O => 3CH_3COOH+NH_3$

9. Methanogenesis
   $CH_3COOH => CH_4+CO_2$

TABLE A2. INITIAL VALUES

| Initial Values | | | | |
|---|---|---|---|---|
| Name | Expression | UNIT | Description | SOURCE |
| C0 | 56.76 | kg/m³ | Initial Carbon Concentration | Measured |
| DBD | 1200 | kg/m³ | Dry Bulk Density of Waste | Measured |
| XI0 | 0.1*Y_ss*$S_s$ | kg/m³ | Initial concentration of aerobic bacteria | (Kim et al., 2007) |
| $S_s0$ | 1 | kg/m³ | Initial Soluble Substrate concentration | Assumed |
| XANOX0 | 5.3146E-4 | kg/m³ | Initial concentration of anoxic bacteria | Assumed |
| XAOB0 | 0.002 | kg/m³ | Initial concentration of AOB | Assumed |
| XNOB0 | 0.002 | kg/m³ | Initial concentration of NOB | Assumed |

TABLE A3. PARAMETERS

| Name | Expression | UNIT | Description | SOURCE |
|---|---|---|---|---|
| bd_a | 7.6E-5 | 1/s | Biomass decay constant for Aerobic bacteria | (Stombaugh and Nokes, 1996) |
| bd_aob | 1.1111E−6 | 1/s | Biomass decay constant for Ammonia Oxidizing Bacteria | (Hiatt and Grady, 2008) |
| bd_nob | 1.1111E−6 | 1/s | Biomass decay constant for Nitrogen Oxidizing Bacteria | (Hiatt and Grady, 2008) |
| DC_N2O | 2.22E-4 | m²/d | N2O diffusion coefficient in water | (Haynes, 2014) |
| DC_N2 | 1.73E-4 | m²/d | $N_2$ diffusion coefficient in water | (Haynes, 2014) |
| DC_NH2OH | 2.09E-4 | m²/d | NH2OH diffusion coefficient in water | (Sabba et al., 2015) |
| DC_NH4 | 1.69E-4 | m²/d | NH4 diffusion coefficient in water | (Haynes, 2014) |
| DC_NO | 2.22E-4 | m²/d | NO diffusion coefficient in water | (Zacharia and Deen, 2005) |
| DC_NO2 | 1.65E-4 | m²/d | NO2 diffusion coefficient in water (Haynes 2014 as seen in Chen 2019) | (Haynes, 2014) |
| DC_NO3 | 1.64E-4 | m²/d | NO3 diffusion coefficient in water (Haynes 2014 as seen in Chen 2019) | (Haynes, 2014) |
| DC_O2 | 1.98E−5 | m²/s | Diffusion Coefficient of O2 in air at 293.15K | (Schmitz et al., 2013) |
| DC_O2S | 1.1574E−6 | m²/s | Effective Diffusion Coefficient of O2 in Waste | (Kim et al., 2007) |
| H_a | −4.6E5 | J/mol | Enthalpy of Aerobic Substrate Degradation | (Kim et al., 2007) |
| kH | 4.9E−7 | 1/s | Substrate Hydrolysis coefficient | (Lin et al., 2008) |
| KN2O_HB | 0.05 | g/m³ | $N_2O$ affinity constant | (Hiatt and Grady, 2008) |
| KNH2OH_AOB | 2.4 | g/m³ | $NH_2OH$ affinity constant | (Ni et al., 2013) |
| KNH3_AOB | 2.4 | g/m³ | $NH_3$ affinity constant | (Ni et al., 2013) |
| KNO_AOB1 | 0.0084 | g/m³ | NO affinity constant 1 | (Ni et al., 2013) |
| KNO_AOB2 | 0.0084 | g-N/m | NO affinity constant 2 | (Ni et al., 2013) |
| KNO_HB | 0.05 | g-N/m | NO affinity constant | (Hiatt and Grady, 2008) |
| KNO2_AOB | 0.14 | g-N/m³ | $NO_2$ affinity constant | (Ni et al., 2013) |

| Name | Expression | UNIT | Description | SOURCE |
|---|---|---|---|---|
| KNO2_HB | 0.2 | g-N/m³ | $NO_2$ affinity constant | (Hiatt and Grady, 2008) |
| KNO2_NOB | 5.5 | g-N/m³ | $NO_2$ affinity constant for NOB | (Wiesmann, 1994) |
| KNO3_HB | 0.2 | g-N/m³ | $NO_3$ affinity constant | (Hiatt and Grady, 2008) |
| Ko | 3.4E-4 | mol/m³ | Oxygen Half saturation coefficient for aerobic bacteria | (Hamelers, 2000) |
| KO2_AOB1 | 0.4 | g-$O_2$/m³ | $O_2$ affinity constant 1 | (Ni et al., 2013) |
| KO2_AOB2 | 0.073 | g-$O_2$/m³ | $O_2$ affinity constant 2 | (Ni et al., 2013) |
| KO2_AOBi | 0.112 | g-$O_2$/m³ | $O_2$ inhibition constant | (Ni et al., 2013) |
| KO2_HB | 0.1 | g-O2/m³ | $O_2$ affinity constant | (Hiatt and Grady, 2008) |
| KO2_i | 0.1 | g-O2/m³ | $O_2$ inhibition constant | (Hiatt and Grady, 2008) |
| KO2_NOB | 2.2[g/m^3] | g-O2/m³ | O2 affinity constant for NOB | (Wiesmann, 1994) |
| Ks | 1 | kg/m³ | Substrate Saturation Constant (Organic substrate respiration) | (Rees-white et al., 2008) |
| Ks_1 | 20 | g-COD/m³ | Substrate affinity constant 1 (for NO3- to NO2-) | (Hiatt and Grady, 2008) |
| Ks_2 | 20 | g-COD/m³ | Substrate affinity constant 2 (for NO2 to NO) | (Hiatt and Grady, 2008) |
| Ks_3 | 20 | g-COD/m³ | Substrate affinity constant 3 (for NO to N2O) | (Hiatt and Grady, 2008) |
| Ks_4 | 40 | g-COD/m³ | Substrate affinity constant 4 (for N2O to N2) | (Hiatt and Grady, 2008) |
| an_1 | 0.28 | - | Anoxic growth factor 1 (for $NO_3$- reduction to $NO_2$ | (Hiatt and Grady, 2008) |
| an_2 | 0.16 | - | Anoxic growth factor 2 (for $NO_2$- reduction to NO) | (Hiatt and Grady, 2008) |
| an_3 | 0.35 | - | Anoxic growth factor 3 (for NO reduction to $N_2O$) | (Hiatt and Grady, 2008) |
| an_4 | 0.35 | - | Anoxic growth factor 4 (for $N_2O$ reduction to $N_2$) | (Hiatt and Grady, 2008) |
| $T_{max}$ | 71.6 | ºC | Maximum temperature for biodegradation | (Richard and Walker, 2006) |
| $T_{min}$ | 5 | ºC | Minimum temperature for biodegradation | (Richard and Walker, 2006) |
| $T_{opt}$ | 58.6 | ºC | Optimal temperature for biodegradation | (Richard and Walker, 2006) |
| μ _aob1 | 4.92 | g-N/(g-COD*d) | Maximum reaction rate for $NH_4$ to $NH_2OH$ | (Ni et al., 2013) |
| μ _aob2 | 2.04 | 1/d | Maximum specific growth rate for $NH_2OH$ oxidation to NO | (Ni et al., 2013) |
| μ _aob3 | 13.61 | g-N/(g-COD*d) | Maximum reaction rate for NO oxidation to $NO_2$ | (Ni et al., 2013) |
| μ _aob4 | 0.58 | g-N/(g-COD*d) | Maximum reaction rate for NO reduction to $N_2O$ | (Ni et al., 2013) |
| μ _aob5 | 0.16 | g-N/(g-COD*d) | Maximum reaction rate for $NO_2$ reduction to $NO_2$ | (Ni et al., 2013) |
| μ _nob | 1.36 | 1/d | Maximum Specific growth rate of Nitrite-oxidizing bacteria | (Volcke et al., 2006) |
| μ max | 1 | 1/d | Maximum specific growth rate for Aerobic bacteria | (Kim et al., 2007) |
| Y_aob | 0.18 | [g-COD/g-N] | Yield coefficient for Ammonia Oxidizing Bacteria | (Hiatt and Grady, 2008) |
| Y_nob | 0.06 | [g-COD/g-N] | Yield coefficient for Nitrate Oxidizing Bacteria | (Hiatt and Grady, 2008) |
| Y_ss | 0.6 | [mol/mol] | Biomass Yield Coefficient on Aerobic Respiration | (Hiatt and Grady, 2008) |

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# CHAPTER 5

# Conclusions and Recommendations

# CHAPTER 5

## 5. Conclusions and Recommendations

### 5.1 Conclusions

Organic landfills have been utilized for many generations and are still the most prevalent form of waste disposal worldwide as developing countries still rely on them due to lower capital costs. As such they will continue to be a part of the solid waste management infrastructure for years to come. If developing countries implement more advanced techniques such as incineration there will still be a need for final disposal sites for the incineration ashes. As such landfills are always needed.

In many developing countries informal landfills are created frequently and can grow large over time if left unchecked. These sites pose a burden to the natural environment as the leachate they generate remains untreated and escapes into the environment. They are also a source of harmful greenhouse gas emissions. This thesis explored and potential method which could be used to remediate informal landfills and allow for their reuse in a shorter period.

The experiments showed that with high levels of available organic matter which is expected to be present in developing country landfills aerobic decomposition of carbon will be promoted and create a positive loop of promoting bacterial growth which may promote further decomposition of nitrogenous substances as well.

From the simulations the system presented here of utilizing passive aeration pipes as a means of aerating existing landfills can provide a possible 60% improvement in the oxygen levels and nitrification process for low cost and low maintenance. The use of passive ventilation appears to show some of the beneficial characteristics of landfill treatment systems such as intermittent aeration or semi aerobic aeration with much less overhead cost and in a system which can potentially be implemented at any preexisting or future landfills.

### 5.2 Recommendations

While the study aimed to study the effect of the introduction of passive ventilation on organic waste masses a conservative the lysimeters were filled conservatively with 10% compost and 90% bottom ash. This is thought to have affected the data collection as the alkaline bottom ash may have reacted with dissolved $CO_2$. This resulted in lower-than-expected $CO_2$ gas readings. For future experiments bottom ash should be replaced with another inert material or compost proportions should be significantly increased.

The study was initially constructed to focus on the main gases of oxygen, nitrogen, carbon dioxide, and methane. No provisions were made or available to include dinitrogen oxide ($N_2O$) in the experiment which is the precursor to Nitrogen gas in the denitrification pathway. Having this additional data would have given greater insight into how the denitrification process was affected by the presence of the pipes. Future lysimeter investigations on passive aeration should also incorporate the study of ($N_2O$) closely into the study.

For future simulations additional factors, such as pH, can be integrated into simulations as pH can be an important control of microbial processes.