“By knowing things that exist, you can know
that which doesn’t exist”

-Miyamoto Musashai

University of Alberta

Greenhouse Gas Emissions from Composting and Stockpiling of Beef Feedlot Manure

by

Gurpreet Singh ©

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Abstract

The study was conducted to compare the greenhouse gas (GHG) (carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)) emissions from two different manure management practices. The two piles, compost and manure stockpile, were established using the same beef feedlot manure. The compost pile was turned frequently while the manure stockpile was static throughout the trial. This study involved the collection of biogenic GHG emissions using static chamber methodology and estimation of non-biogenic GHG emissions during the anthropogenic activities. Carbon dioxide, methane and nitrous oxide emissions were significantly ($P \leq 0.05$) different from two treatments. The total GHG emissions from the compost pile and the manure stockpile were estimated to be 397 and 364 kg CO_2-equivalent Mg^{-1} initial dry mass. However non-biogenic GHG emissions from the two treatments were negligible as compared to biogenic emissions. This project will enhance the prediction of GHG emissions from compost and manure stockpiling and may form the basis to extend this study further for long term analysis.

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List of Symbols and Abbreviations

A_j	Cross-sectional area of pile (m^2)
A_s	Chamber basal area (m^2)
AAFC	Agriculture and Agri-Food Canada
AAFRD	Alberta Agriculture, Food, and Rural Development
AFNS	Agricultural Food, and Nutritional Science Department
B.D.	Bulk density (kg m^{-3})
B.Q.	Fuel Consumption (L)
$\dot{C}$	Rate of change in concentration (μL L^{-1} s^{-1})
CCME	Canadian Council of Ministers of the Environment
E	Efficiency (%)
ECD	Electron capture detector
EF	Emissions factor (g-GHG/amount of fuel burned)
EMS	Earthen manure storage
$\dot{f}$	Mass flux per unit area (g s^{-1} m^{-2})
FAO	Food and Agricultural Organization of the United Nations
FAS	Free air space (%)
FID	Flame ionization detector
GC	Gas chromatograph
GHG	Greenhouse gas
GWP	Global warming potential
H_x	Area under the curve (m^2)
HFC	Hydroflourocarbon
i	Index of measurements in a cross-section (i = 1, 2,.....,216)
IPCC	Intergovernmental Panel on Climate Change
IR	Infra red
j	Index of cross-sections along the length (j = 1,2,….,12)
L	Length of pile (m)
M	Molar mass (g mol^{-1})
$\dot{m}$	Chamber mass flux (g s^{-1})
M.C.	Moisture content (%)
N	Number of Moles
OM	Organic matter
P	Pressure (kPa)
ρ	Density (kg m^{-3})
PVC	Polyvinyl chloride
R	Universal gas constant (0.08207 L atm mol^{-1} K^{-1})
T	Temperature (^{o}C)
TC	Total carbon (%)
TN	Total nitrogen (%)
V	Volume (m^3)
v_m	Molar volume of an ideal gas (22.4 L mol^{-1})

V_s	Volume of static chamber (3.3 L)
W	Total width of pile (m)
Δx	Incremental width of pile (m)
Δy	Incremental length of pile (m)

Chapter 1. Introduction

1.1 BACKGROUND

Atmospheric concentrations of greenhouse gases (GHGs) have increased since the industrial era started in 1750. For instance, carbon dioxide (CO_2) concentration has risen by 30%, methane (CH_4) concentration has doubled, and nitrous oxide (N_2O) concentration has increased by about 15 % (IPCC 1995). Greenhouse gases act to trap long-wave radiation emitted from the Earth's surface.. The ability of each GHG to trap radiation is termed global warming potential (GWP). The GWPs of CO_2, CH_4, and N_2O are 1, 23, and 296, respectively (IPCC 2001a). These GHGs are believed to accelerate the warming of the atmosphere and influence climate change, which is a significant international concern and is the subject of much research and debate. According to the Intergovernmental Panel on Climate Change (IPCC 1996), the Earth's surface temperature has increased dramatically about 0.2°C per decade since 1975 due to global warming. Today, society faces important decisions regarding climate change mitigation and concerns are being expressed about the potential impact of the build up in atmospheric concentrations of GHGs. A scientific consensus is emerging that this build up will affect the global climate, most likely, stimulating warming. The IPCC argues that it will take: a) centuries for the sea level to stop rising from a temperature increase; b) decades for atmospheric GHG concentrations to stabilize even after emissions have stabilized (IPCC 1996).

In Kyoto, December 1997, 190 countries participated in a conference to discuss issues concerning global warming due to GHG emissions. As a result of the negotiations, the Kyoto Protocol came into existence, which committed signatory industrialized nations to reduce emissions of GHGs to 5.2% below 1990 levels. Canada was among 174 nations that signed the Kyoto Protocol. Canada agreed to lower its emissions 6% below 1990 levels over a span of five years from 2008 to 2012 (Desjardins *et al.* 2001). It is noteworthy that in 2003 Canada contributed only about 2% of the total global anthropogenic GHG emissions, but GHG emissions were among the highest per capita (Environment Canada 2003). In order to achieve the reduction target set out in the Kyoto

Protocol, all provincial governments in Canada have been asked to explore key GHG-emitting sectors and, subsequently, to work towards the implementation of the protocol.

Apart from the key GHG-emitting areas, such as the industry and transportation sectors, the agriculture and waste management sectors have also been identified as major contributors of GHGs, accounting for 11.2% of total GHG emissions in Canada in 2000 (Environment Canada 2000). According to Janzen *et al.* (1998), animal manure management was estimated to be the source of 1.25% of GHG emissions in Canada in 1998. Manure storage mainly releases CO_2, CH_4, and N_2O gases during decomposition. In 2002, manure in Canada released about 270 and 15 kt of CH_4 and N_2O, respectively (Environment Canada 2003).

The ratification of the Kyoto Protocol by the Canadian government has raised interest in the development of improved waste management practices that would reduce GHG emissions. Composting of organic waste is one of several potentially valuable options available to manage animal manure. Composting is gaining acceptance as an alternative to conventional methods of manure management due to associated benefits such as easy handling, and reduction in volume of organic waste (Hao *et al.* 2001). It is generally believed that composting of organic waste also reduces GHG emissions compared to conventional systems in which manure is simply stockpiled before being spread on the land.

Previous research on composting has been focused mainly on "composting as an effective tool for waste management" (Zeman *et al.* 2002). Very little data are currently available which quantify the total GHG emissions from composting. There are still large uncertainty and variability in estimates of GHG emissions from composting (He *et al.* 2000). Although it is sometimes assumed that only CO_2 is produced during composting, this is only applicable when aerobic conditions are strictly maintained throughout the entire composting process. Several previous studies have reported, however, that CH_4 and N_2O are also produced along with CO_2 during normal composting of livestock waste (Hao *et al.* 2001; Beck-Friis *et al.* 2000; He *et al.* 2000).

Greenhouse gas emissions from composting and manure stockpiles can be categorized into biogenic (non-anthropogenic) and non-biogenic (anthropogenic) emissions. Biogenic emissions arise from natural decomposition of the organic matter present in the pile. Non-biogenic emission results from the burning of fossil fuel used in the machinery required to perform different operations, from the formation of piles to the spreading of the finished product. Most previous studies have only looked at biogenic emissions from compost or manure, but it is also important to account for non-biogenic emissions from manure management systems.

1.2 OBJECTIVES

The main objective of this research was to quantify and compare the GHG emissions from two manure management processes: composting and manure stockpiling prior to field spreading of the end-product. In this project, an integrated assessment of GHG emissions from both biogenic and non-biogenic sources was attempted. Figure 1.1 illustrates the system boundaries for this study.

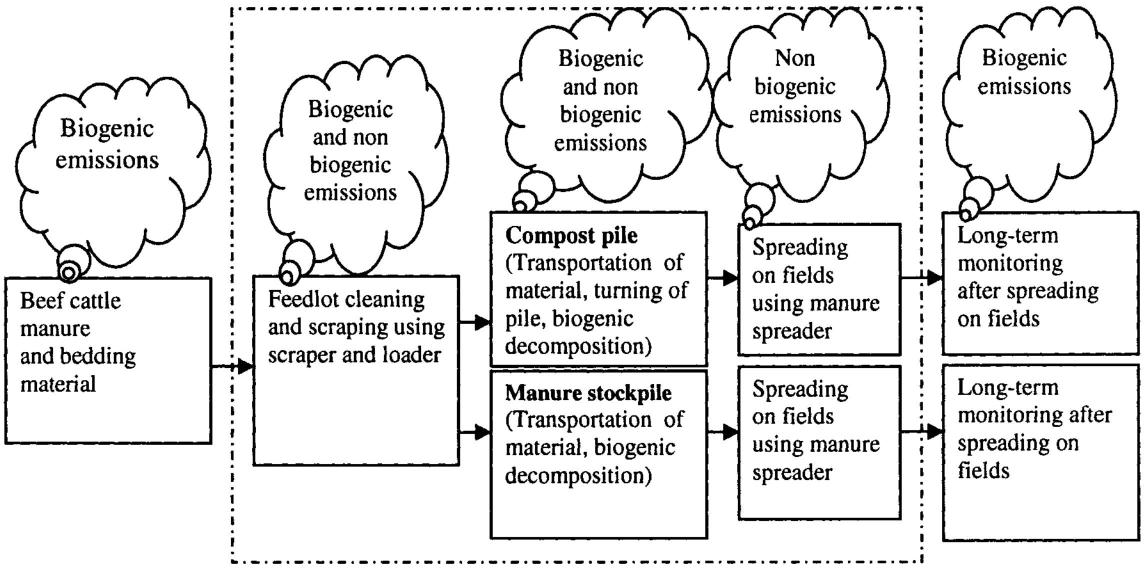


Figure 1.1 Schematic of unit operations and material flow in manure composting and stockpiling (dotted lines represent system boundary)

The following specific goals were identified in this research:

- Measurement of biogenic GHG emissions from active decomposition of beef cattle manure in composting and manure stockpiling.
- Estimation of non-biogenic GHG emissions produced from various machinery operations in composting and manure stockpiling.
- Comparison of total GHG emissions from composting and manure stockpiling.

1.3 OUTLINE OF THESIS

This chapter (Chapter 1) is an introduction and statement of the objectives of the research. Chapter 2 is a literature review of composting and manure stockpiling and factors affecting GHG emissions from these processes. Chapter 3 is a description of the methodology and materials used in conducting this research. Chapter 4 is a presentation of the results and a discussion of the study. Chapter 5 includes the conclusions from the study and recommendations for further research. Chapter 6 is a list of the references used in this thesis.

Chapter 2. Literature Review

2.1 INTRODUCTION

This chapter is an overview of the previous research work done on composting and manure stockpiling and the factors affecting GHG emissions from these processes.

2.2 MANURE MANAGEMENT

Manure management, as it relates to the livestock industry, is described as the collection, storage and disposal of animal waste (AAFRD 2004a). Due to the cold climate of Alberta, livestock housing systems are designed to maintain temperature, humidity, and gaseous contaminants within acceptable ranges. The design of housing systems depends on manure type and the handing system. Livestock manure can be classified as liquid, semi-solid, or solid manures. Liquid manure typically contains 5-10% solids. The manure is removed by a high volume water flush, a continuous flow, or a mechanical scraper. Washwater is added to manure to reduce its solid content. Semi-solid manure contains 10-20% total solids, resulting from bedding material, and washwater is not included in semi-solid manures. Manure is moved using a scraper or tractor blade and is stored on a concrete pad with concrete sides. Washwater is pumped into a separate storage pit. Solid manure usually contains more than 20% total solids content. Depending on the size of the operation, solid manure is cleaned out several times per year. The solid manure is usually stockpiled or is spread immediately.

Manure can be stored either aerobically or anaerobically. Aerobic storage is manure storage in the presence of oxygen while anaerobic storage is in the absence of oxygen. Aerobic storage can be achieved by the mechanical mixing of liquid or semi-liquid manure or the mechanical turning of solid manure. The storage of liquid manure in unaerated lagoons and solid manure in stockpiles without turning is considered as anaerobic storage (AAFRD 2004). In Alberta, manure is stored mostly under anaerobic conditions in one of three ways: a) liquid storage in an earthen manure storage (EMS) (most common type of storage in Alberta); b) semi-solid storage including a minimal amount of bedding material; c) solid storage including a thick layer of bedding material.

2.2.1 Manure handling in beef feedlots

In a typical open beef feedlot, as in this project, manure is left in the pen for more than three months before removal. Usually cattle are kept on a solid floor thickly bedded locally available dry amendments (e.g. cereal, wheat straw). The beef manure is removed by a “push” or “pull” blade mounted on a tractor 2-3 times in a year depending on the cattle count and volume of manure. The excess manure from the feedlot is temporarily stored in either mounds or in an EMS before its final disposal as a field application. However, these traditional manure management and storage techniques such as EMS and spreading of manure on agricultural fields can be associated with environmental concerns such as ground and surface water pollution and odour emissions. Studies have suggested that anaerobic manure management and storage techniques are one of the greatest agricultural threats to the environment (AAFRD 2004). As part of the ongoing effort to develop new, sustainable, and environmentally friendly strategies in the field of manure management, composting has been proposed an alternative to traditional manure management and storage techniques (DeLuca and Deluca 1997). It is generally believed that composting of organic waste also reduces GHG emissions compared to conventional systems in which manure is simply stockpiled before being spread on the land.

2.3 DEFINITION OF COMPOSTING

Different authors have given different definitions of composting and no single definition is universally accepted. In general, however, composting can be defined as the natural process of decomposition of organic matter by microorganisms under controlled conditions (FAO 1980). Composting has been practiced for centuries. Farmers in the early eighteenth century were practicing natural composting (Rynk 1992), but methods of composting have evolved. With the onset of the industrial era, scientific principles were applied to composting. Selective materials, mechanical devices, and alternative methods of composting were introduced to speed up the natural decomposition of the organic matter.

The purpose of composting is biological decomposition and the stabilization of the organic substrate, under conditions that allow the development of thermophilic temperatures (>60°C) as a result of biologically produced heat, to produce a final product that is stable, free of pathogens and plant seeds, and can be beneficially applied to land (Haug 1995; Leonard 2001). During composting, the microorganisms consume oxygen (O_2) while feeding on organic matter and generating heat, and large quantities of CO_2 and H_2O. The basic composting process can be illustrated by Figure 2.1.

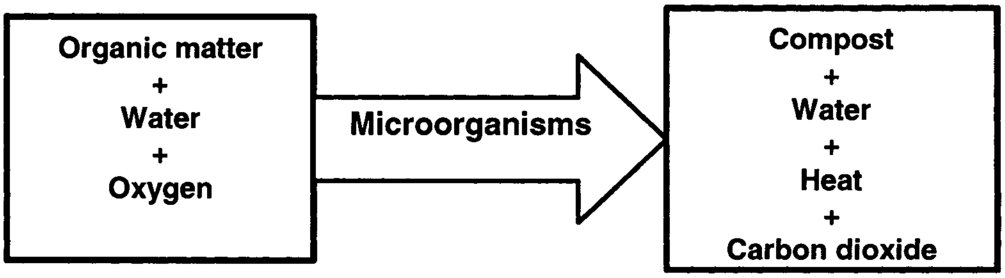


Figure 2.1 Basic composting process

2.4 MICROBIOLOGY OF COMPOSTING

Composting occurs through the efforts of organisms called 'decomposers', which are ubiquitous (Rynk 1992). Different types of micro and macro-organisms, such as bacteria, fungi, actinomycetes, protozoa, mites, millipedes, centipedes, sowbugs, snails, spider, springtails, beetles, ants, flies, worms, and earthworms can be present during the decomposition of organic material. These organisms play an important role in the decomposition of the organic matter present in waste. The most common functional groups of microorganisms found in compost are bacteria, fungi, yeasts, and actinomycetes (Herrmann and Shann 1997).

2.4.1 Bacteria

Bacteria are the smallest, simplest free living organisms. Bacteria are typically unicellular, but multicellular association of individual cells is also known (Rynk 1992). They exist in a wide variety of forms and in a wide range of environmental conditions. The most common morphological forms are spheres (coccus), rods (bacillus), and spirals (spirillum) (Debertoldi *et al.* 1996). Because of their small size, bacteria have a very high surface to volume ratio. The higher surface to volume ratio allows for rapid transfer of soluble substrate into the cells and results in a high rate of metabolic activities (Haug

1995). Bacteria can be classified into three different types according to their metabolism: Aerobic bacteria thrive in the presence of O_2 and require it for growth. Anaerobic bacteria can live without oxygen. Facultative bacteria can thrive under both aerobic and anaerobic conditions. Bacteria are the most metabolically diverse of all organisms present in compost (Shilesky and Maniotis, 1969; Poincelot 1977).

2.4.2 Fungi

As compared with bacteria, fungi are large microorganisms with a eukaryotic cell structure (Rynk 1992). They form networks of individual cells in strands or filaments. Most fungi are saprophytes, feeding on dead or decaying material. They are more tolerant of low moisture and low pH conditions than bacteria but are less tolerant of low O_2 environments. Fungi mostly decompose woody substances and decay resistant materials. At temperatures above 60°C, fungi and yeasts disappear almost totally from compost (Rosenberg 1975). Fungi become more dominant at the end of the composting process, during cooling and maturation of the compost, when the moisture content and pH of the organic matter decrease (Nakasaki *et al.* 1985).

2.4.3 Actinomycetes

Actinomycetes form filaments like fungi but, because of their small size and prokaryotic cell structure, they are referred as mold-like bacteria (Rynk 1992). Actinomycetes are the primary decomposers of tough materials like cellulose and lignin. They are more pronounced after the easily degraded compounds are gone and when moisture levels are low. They appear as a blue-gray to light green powdery or filamentous layer in the pile that gives a pleasing earthy smell to compost. They are often found in drier parts of the pile and can survive a wide range of temperature.

2.5 TEMPERATURE EFFECT ON MICROBIAL ACTIVITY

Naturally occurring microorganisms are temperature sensitive. Temperature is a universal determinant of metabolic activity and it has been recognized as a key environmental factor with a consistent and dramatic effect on microbial activity (Finstein and Hogan 1993; Mckinley and Vestal 1985). According to Ishii *et al.* (2000), temperature evolution in composting can be divided into four different phases: mesophilic, thermophilic, cooling, and maturation. Other researchers, however categorize the active degradation of organic matter into only two phases: An initial and final phase at lower temperature (25 to 40°C) called the mesophilic zone and a intermediate phase of composting corresponding to higher temperature range at (40 to70 °C) called the thermophilic zone (Gray *et al.* 1971; Bagstaun 1978). Figure 2.2 shows these temperature zones. Thermophilic temperature is desirable for rapid decomposition. To destroy pathogens and weed seeds, temperature must stay above 55°C for at least three days (Leonard 2001; Grundy *et al.* 1998). Hess *et al.* (2004) carried out a laboratory-scale study to find the relationship between temperature and *Escherichia coli (E. coli)* pathogen survival during manure composting. The experiments were conducted in bioreactors operated in three temperature ranges: 40 to 50 °C, 50 to 60 °C, and greater than 60°C. The researchers observed an initial decline of *E. coli* with subsequent regrowth when compost temperature did not rise above 50°C. They concluded that the temperature must be maintained at or above 55°C for at least three days during in-vessel composting, and for 15 days during windrow composting.

Noble and Roberta (2004) reviewed several previous studies on the eradication of plant pathogens and nematodes and found that a temperature of 64 to 70°C during 21 days was sufficient to reduce pathogen numbers below detection limits for 33 out of 38 fungi, oomycetes, bacteria, nematodes, and plant viruses.

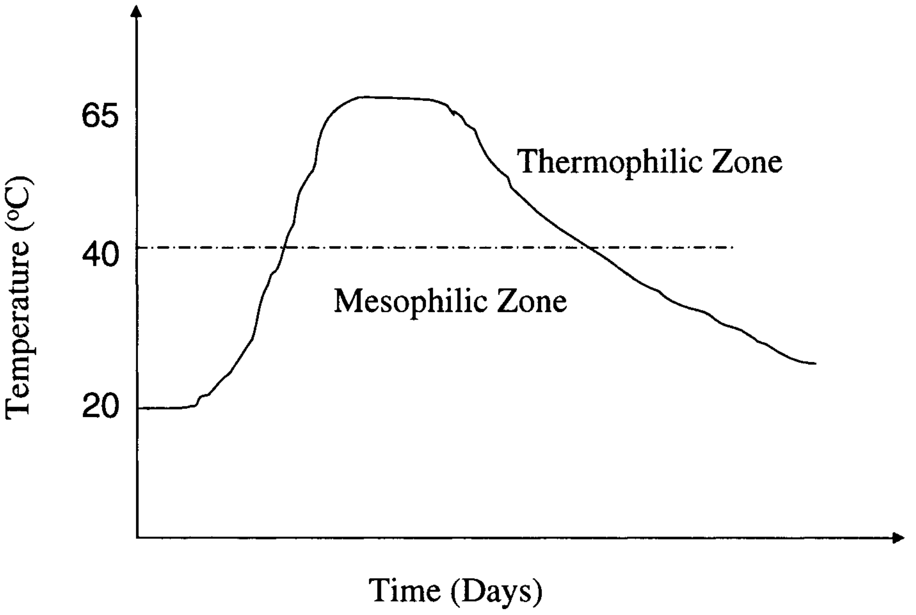


Figure 2.2 Temperature regimes during composting

(Adapted from: Leonard, 2001)

A study conducted by Golueke (1992) on bacteriology of composting found that the efficiency of the composting in terms of microbial population drops at the higher thermophilic levels above 76°C. The author suggested that temperature should not be allowed to exceed 55–60°C. Conversely, a few studies reported high numbers of viable microorganisms and high cellulose degradation at 60–75°C (Poincelot 1974; Nakasaki *et al.* 1986; Stenbro-Olsen 1998).

Ryckeboer *et al.* (2003) conducted a comprehensive study on the microbiological aspects of biowaste during composting in a monitored compost bin. They found that temperature clearly influenced microorganism populations and bacterial activity. Different bacterial populations were observed in the thermophilic and mesophilic phases; however bacteria were the dominant functional group during the thermophilic phase. Cellulolytic bacteria were found to be present even during peak heating while fungi and actinomycetes were below the detection limit. After cooling (<45°C), the population of bacteria increased at least 100-fold compared to the population observed during the thermophilic phase. Fungi populations were observed after the cooling of compost (<45°C) and were active until the end of composting. The researchers concluded that the high temperatures (>70°C) during the thermophilic composting phase resulted in a decline of the total microbial population. They also concluded that fungi and actinomycetes proliferated only when the temperature was low (<45°C).

2.6 FACTORS AFFECTING THE COMPOSTING PROCESS

Composting is a natural biological process. However, natural processes can be accelerated by controlling physical and chemical factors. The physical factors include moisture content, bulk density, porosity, particle size and oxygen concentration. Chemical factors include carbon to nitrogen ratio (C:N), pH, and temperature. All of these factors affect the performance of the composting process (Rynk 1992).

2.6.1 Physical factors

- **Moisture content**

Water is necessary for all living organisms, including microorganisms, to support the metabolic processes. Water provides the medium for chemical reactions and transport of nutrients (Rynk 1992). On one hand, microorganisms will not be active if the moisture content (M.C.) is too low and, on the other hand, if M.C. is too high, water can displace the air from the pore spaces, resulting in anaerobic regions within the pile (Haug 1995). Excessive water not only fills available pore spaces that would result in leachate production. According to Miller (1991), the optimum M.C. is a trade-off between providing the moisture necessary for the microbial communities to flourish while still maintaining sufficient oxygen and porosity within the pile. However, the oven-drying technique is the most widely used methods for measuring compost M.C.

Several previous studies reported different optimum ranges of M.C. for effective composting process. Rynk (1992) suggested that 40-65% M.C. works well for effective decomposition of organic matter. Even though M.C. at the initial stages of composting may be optimum, moisture is evaporated due to turning of the pile and heat generation by microbes inside the pile. It has been suggested the initial M.C. of the manure be close to upper limit of 65%. Liao *et al.* (1993) successfully composted swine manure solids mixed with sawdust in a 5:1 ratio by weight with an overall M.C. of 71%.

Richard *et al.* (2002) described moisture as a key process variable because of the direct effect of moisture on material and matrix properties and, hence on microbial activity. The results of their study suggested that the optimum moisture content for biodegradation can vary for different compost types and materials, and range from 50-70% on a wet basis. They found a significant reduction in the biodegradation rate when the M.C was outside the optimum range of 50-70%. They concluded that composting might be feasible even at higher M.C. as long as sufficient porosity inside the pile is maintained to satisfy the oxygen needs of microbes.

- **Relative water-filled pore space**

The concept of water-filled pore space (WFPS), which uses bulk density and porosity values, is defined as amount of water present in the available pore spaces (Stanford and Epstein, 1974; Linn and Doran, 1984).

Franzluebbers A.J. (1999) has suggested an equation to calculate WFPS as:

$$WFPC = (WCxBD)/(1-(BD/PD))$$

Where,

WFPC is water filled pore space (m^3 water m^{-3} pore space), WC is water content ($g\ g^{-1}$), BD is the bulk density ($Mg\ m^{-3}$), and PD is the particle density ($Mg\ m^{-3}$).

In manure management practices, most researchers are using simple mass per unit volume technique to estimate bulk density. Agnew and Leonard (2003) have suggested an air pycnometer method to calculate porosity of the compost however, in this study porosity of the material was not analyzed.

- **Bulk density**

The bulk density (B.D) of compost is a measure of the mass of material within a given volume. It determines in part the volume of manure that can be spread at a certain site (Agnew and Leonard 2003). Bulk density is a key physical parameter of compost which influences mechanical properties such as strength, porosity, and ease of compaction (Rynk 1992; Larney *et al.* 2000).

According to He *et al.* (1995), the majority of compost researchers use a simple mass per unit volume technique to estimate B.D. A container of known mass and volume is filled with material and is slightly compacted to ensure absence of large void spaces. The weight of the material and container is then measured, and the bulk density can be calculated. Leege and Thompson (1997) slightly modified this method and recommended that a partially filled container be dropped from 100 mm onto a rubber mat. This settles the material and eliminates large pores present in the material. Agnew and Leonard (2003) in their research also emphasized the need of a standardized method to determine bulk density.

- **Aeration requirement**

The most rapid decomposition of organic matter during composting is done by aerobic microorganisms, which require sufficient amounts of O_2 to decompose the raw material (Rynk 1992). The oxygen demand of the microorganisms is greatest during the initial stages of composting. A minimum of 5% O_2 concentration is necessary for effective and odorless aerobic composting (Brodie *et al.* 2000).

2.6.2 Chemical factors

- **Carbon to nitrogen ratio**

Carbon (C) and nitrogen (N) compounds can inhibit the composting process if they are present in either excessive or insufficient amounts. Microorganisms in organic matter metabolize C compounds as an energy source while N is an essential element for protein synthesis. Efficient decomposition occurs when the proportion of these two elements in the range of 20 to 40:1 (Haug 1995). The C:N ratio of selective materials are shown in Table 2.1.

Table 2.1 Carbon-to-nitrogen ratio of selected material

Material with high N value	**C:N**
Poultry manure with litter	13-18:1
Vegetable waste	12-20:1
Pig manure solids	15-25:1
Dairy manure	20-25:1
Material with high C value	
Wood chip and saw dust	100-500:1
Paper waste	15-200:1
Straw waste	40-100:1
Foliage	30-80:1
Material with neutral C:N ratio	
Peat moss	18-36:1
Horse manure with litter	30-60:1

Adapted from Rynk (1992)

Green waste and fresh manure are rich in N with high M.C. around (>80%) while sawdust, wood shavings, and straw are good sources of C. At an optimum C:N ratio, microorganisms decompose organic matter quickly. A low C:N ratio (<10:1) results in the release of surplus nitrogen into the atmosphere in the form of ammonia (NH_3), posing an odor problem. Decomposition remains incomplete at a high C:N ratio because of under-utilization of C (Pare *et al.* 1998).

Several previous studies analyzed the effect of the C:N ratio on the decomposition of organic matter and reported a decrease in the C:N ratio from the initial value during composting. Baddi *et al.* (2003), in their chemical and spectroscopic analyses of organic matter transformations during composting of olive mill wastes, observed a 160% decrease in the C:N ratio after 12 months of composting. The initial C:N ratio (69:1) decreased to 16.9:1. Similarly Levanon *et al.* (2002) examined physical, chemical, and biological criteria of dairy manure compost found a C:N ratio decrease to 12.5:1 from an initial value of 22:1 over a seven months of composting.

Sadaka and El-Taweel (2003) carried out a study on household waste composting to determine the optimum aeration level and C:N ratio for achieving thermophilic temperature (>68°C). Three different air flow levels (0.001, 0.003, and 0.006 $m^3 h^{-1}$) and three different C:N ratios (11, 26, and 39) were studied in a laboratory scale composting unit. They concluded that compost reached a maximum temperature (68°C) at an aeration level of 0.003 $m^3 h^{-1}$ and C:N ratio of 26.

- **pH of material**

Composting may proceed effectively over a wide range of pH without seriously limiting the process. The optimum range of pH, however, is 6.5-7.5 for microorganisms involved in composting (Rynk 1992).

Elwell *et al.* (2001), in their study on odorous emissions and odor control in composting swine manure/sawdust mixes, found that the initial pH influenced the rate of increase of bacterial activity in the swine manure–sawdust mixes. A pH of 6.0 appeared to be a critical lower boundary. Below this value, temperature rise was retarded and the maximum temperature varied between 30 and 35°C. These effects indicated relatively slow growth of bacterial populations restricted to mesophilic activity.

The pH is an important consideration with raw material that has a high nitrogen value. A high pH (>8.5) encourages the conversion of nitrogen to ammonia, which can lead to odor problems (Liang *et al.* 2004). Roig *et al.* (2004) carried out lab and field scale studies to control pH during the composting of olive mill waste, using elemental sulphur (S) as an alternative to lime. Sulphur is oxidized to H_2SO_4 by sulphur-oxidizing microorganisms. Sulphur was used at a rate of 0.5% on dry mass basis. They found that S oxidation took place mainly during the first 10 days of incubation and pH was reduced by nearly two units. It was concluded that an increase in S addition led to a greater decrease in pH, but the addition of large amounts of S would have negative effects on microbial populations.

2.7 COMPOSTING METHODS

2.7.1 Passive windrow composting

Passive windrow composting is the production of compost in the piles with little management. No aeration is provided by external sources and the compost is produced by natural aeration over a long period of time. Decomposition is slow because of low aeration.

2.7.2 Turned windrow composting

In turned windrow composting, aeration is provided by turning the pile using a mechanical turner. Turning helps to mix and pulverize materials, breaking up the particles into smaller sizes, restoring the porosity of the windrow, and releasing trapped heat, water vapor and gases. The rate of air exchange depends on turning frequency and the porosity of the windrows.

2.7.3 Passively aerated windrows

In this method of composting, turning is eliminated. Aeration is supplied through pipes embedded underneath the windrows. Air flows into the pipes and moves upward through the windrow from drilled holes in the pipe due to the chimney effect.

2.7.4 Aerated static pile composting

The basic principle in this method is the provision of air by mechanical aeration. Aeration is achieved by blowing or drawing air through the composting material. A typical aeration system includes a blower/fan, electrical motors, ducts, etc. Aeration requirements depend on the type of material to be composted and the volume of material in the windrows. For effective aeration through the compost pile, the material should also be adequately porous. Proper initial mixing of the compost material with amendments is therefore an important consideration.

2.7.5 In-vessel composting

In-vessel composting is defined as composting in drums, silos, containers, or channels using precise control of composting parameters. In-vessel composting generally takes place inside a building using a forced aeration system and mechanical turning of the compost. The advantages of in-vessel composting over other composting methods are high process efficiency, close monitoring of composting parameters, and elimination of odor problems.

2.8 GREENHOUSE GASES

The interest of the world community has raised the implementation of the Kyoto Protocol in understanding the greenhouse effect and ways of reducing GHGs. Greenhouse gases act to trap long-wave radiation emitted from the Earth's surface, increasing the warming of the atmosphere and influencing climate change (IPCC 2001). The major GHGs are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). The atmospheric concentrations of CO_2, CH_4, and N_2O have increased by 30%, 100%, and 15%, respectively, over estimated pre-industrial levels, i.e. before 1750 A.D. (IPCC 2001). These significant increases in concentration are due to the use of fossil fuels and other anthropogenic (human) activities. Direct combustion and non-energy uses of fossil fuels are responsible for about 98% of total anthropogenic CO_2 emissions and 78% of total anthropogenic GHG emissions (IPCC 2001).

2.8.1 Greenhouse effect

The Earth receives radiant energy from the sun but that energy is re-radiated back into space, so that the global temperature remains approximately stable. Solar energy arrives at the Earth's surface at wavelengths lying predominantly within the visible part of the electromagnetic spectrum. However, the Earth re-radiates energy at longer wavelengths in the infrared or thermal region of the spectrum. Energy at these longer wavelengths is more readily absorbed by CO_2 and water vapor, in addition to other infrared-absorbing gases such as CH_4, N_2O, chlorofluorocarbons (CFC), and ozone (O_3). This absorption occurs primarily in the troposphere, the atmospheric region from the Earth's surface up to an altitude of 10 to 15 km. When molecules of these gases absorb energy, they cause general atmospheric warming. These gases thus act like a "thermal blanket" around the Earth. With the increase in the concentration of (GHGs) in the atmosphere, the equilibrium temperature of the atmosphere increases. This phenomenon is called the "greenhouse effect" (AAFRD 2004).

The IPCC (1996) has developed an index called the 'global warming potential' (GWP) to compare the ability of each GHG to trap heat in the atmosphere. The GWP of each gas depends on its capacity to absorb and re-emit radiation and on the time period that the gas remains in the atmosphere. The GWP of a GHG is defined as the ratio of the time-integrated radiative forcing resulting from the instantaneous release of 1 kg of a trace substance relative to that of 1 kg of a reference gas (IPCC, 2001). The reference gas used is CO_2 and GWP weighted emissions are measured in terms of CO_2 equivalent. The lifetimes of different gases vary, and the GWP depends on the integration time interval. The most common time interval is 100 years (IPCC, 2001). Table 2.2 lists the GWP values of different GHGs.

Table 2.2 Global warming potentials (100 year time horizon)

Gas	Formula	GWP
Carbon dioxide	CO_2	1
Methane	CH_2	23
Nitrous oxide	N_2O	296
HFC-23	CHF_3	12,000
HFC-32	CH_2F_2	550
Perfluoromethane	CF_4	5700
Sulphur hexafluoride	SF_6	22,200
Perfluoromethane	CF_4	9,000

Adapted from (IPCC 2001)

Carbon dioxide, methane, and nitrous oxide are the three important GHGs emitted from waste management. The primary anthropogenic sources of CO_2 are fossil fuel combustion, deforestation (removal of forest cover), and industrial processes. The primary natural sources of CO_2 generation include respiration by plants and animals, decomposing organic matter and fermentation, volcanoes, forest/grass fires, and the oceans.

Methane is a more reactive and radiatively-active GHG than CO_2 on a molecule basis (21 times) (IPCC 1996). The average lifetime of CH_4 is very short compared to other GHGs: about 9-15 years. The predominant anthropogenic sources of CH_4 are livestock and rice cultivation, coal mining, biomass burning, natural gas delivery systems, and waste management. Methane emissions in waste management arise mostly from anaerobic decomposition of organic mater by microorganisms (Thompson *et al.* 1992).

It has been estimated that approximately one-third of global atmospheric N_2O is of human origin, resulting primarily from the application of nitrogen fertilizers, soil cultivation, and the combustion of fossil fuels and wood. The other two-thirds of global atmospheric N_2O come from soil and water denitrification under anaerobic and aerobic conditions (IPCC 2001). Nitrous oxide mainly leaks from microbial cells during nitrification and denitrification (Firestone and David 1989). In nitrification, ammonium (NH_4^+) is oxidized to nitrate mainly under aerobic conditions in order to derive energy. In denitrification, nitrate is reduced to N_2 mainly by heterotrophic bacteria under oxygen-restricted conditions. However, the complete mechanism of N_2O production by nitrification and denitrification is still not completely understood (Blackmer *et al.* 1980; Stevens and Laughlin 1998).

2.8.2 Biogenic greenhouse gas emissions from manure management

Biodegradation of the organic fraction of waste generates GHGs (He *et al.* 2000). Even if predominantly aerobic conditions are maintained in composting, anaerobic decomposition still occurs inside the waste particles. It was reported in several previous studies that GHGs such as CO_2, CH_4, and N_2O are emitted during composting (Sommer and Moller 2000; Hao *et al.* 2001).

Ballestero and Douglas (1996) conducted a comparison of nitrogen (N_2) losses between farm waste composting and yard waste composting. The authors found that N_2 losses were mainly dependent on the type of organic waste, the amount of carbon present, bulk density, moisture content, and the presence of inorganic material. In their study, N_2O emissions were compared from two composting systems. During the first 30 days of composting, N_2O fluxes from both the farm and yard waste windrows were found to increase at the same rate. After 30 days of composting, N_2O fluxes from the yard waste increased to 2750 mg N_2O-N m^{-2} d^{-1} and remained above 1800 mg N_2O-N m^{-2} d^{-1} until the end of composting, whereas N_2O fluxes from the farm waste remained low, at about 500 mg N_2O- N m^{-2} d^{-1}. The authors concluded that N losses were resulted due to inhibition of oxygen diffusion resulted from higher waste bulk density.

Czepiel *et al.* (1996) conducted a study to measure N_2O emissions from the composting of organic waste. The authors described N_2O emissions as an important phenomenon during livestock waste and wastewater sludge composting. The livestock waste was composted by forming two windrows composed of 25% horse and cattle manure in equal proportion and 75% bedding material. The windrows were turned regularly with a mechanical turner for proper mixing and aeration. Gas samples were collected using closed static chambers and analyzed with a gas chromatograph. Czepiel *et al.* (1996) found that the N_2O emissions increased significantly during 35-55 days of composting. The N_2O emissions were observed at low levels during the first four weeks of composting. The peak N_2O emissions from two livestock piles were 1.3 and 0.8 g m^{-2} day^{-1}. The average N_2O emissions from both piles were 500 mg m^{-2} day^{-1}.

Lopez-Real and Bapista (1996) studied three different composting systems and their effect on CH_4 emissions from cattle manure amended with straw. The manure was managed with three different techniques: 1) minimal intervention, i.e. stockpiling of manure without any disturbance; 2) windrow formation with frequent mechanical turning; 3) forced aeration. Gas samples were collected from static chambers placed on top of the piles. The highest CH_4 concentration (48.6 ppm) was found from the minimal intervention pile on Day 14 of composting. Methane concentrations for the turned and forced aerated piles varied from 2.98 to 9.27 ppm and 0.96 to 5.95 ppm, respectively. The results indicated that forced aeration lowered CH_4 concentrations the most.

Hellmann *et al.* (1997) carried out studies on emissions of GHGs from open-windrow composting of 40% municipal waste and 60% brush, grass clippings and leaves. The windrow was turned twice a week and was covered with fleece fabric to retain moisture. Gas sampling was done using a closed chamber. The author found high CO_2 emission rates (peak value 360 mg Mg^{-1} h^{-1}) until Day 6 of composting. After Day 6, CO_2 emissions decreased sharply and varied between 10 to 100 mg Mg^{-1} h^{-1}. The authors attributed the high initial increase in CO_2 emission rates to the degradation of easily available nutrients, favorable aerobic conditions, and high microbial activity due to high temperature. Nitrous oxide emissions were high during the first day of composting and a sudden increase in the N_2O emission rate was also observed on Day 36. The authors attributed the high N_2O emissions on those specific days to low temperature, which promote nitrification and denitrification. Nitrifying and denitrifying microorganisms are generally non-thermophilic. During the thermophilic stage, very low levels of N_2O emission rate were measured. High CH_4 emissions were observed between Days 10 and 30 and were attributed to increased temperatures. The high temperature (>70°C) stimulated methanogenic microbial activity.

Hellebrand (1998) studied emissions of N_2O and other trace gases during the composting of green waste and grass. During the first trial, wooden boxes (4m x 2m x 1.75m) were filled with alternative layers of grass cutting (0.25 m), soil (50 mm) and solid manure (150 mm). Gas samples were drawn through polythene pipes embedded in the compost to a multi-gas monitor. The highest methane concentration (580 ppm) was observed in grass clippings amended with manure on the third day of sampling whereas CH_4 concentration in grass clipping composting amended with soil decreased with composting time. The author reported that CH_4 concentration occurred due to development of anaerobic regions inside the pile resulting from insufficient aeration. Nitrous oxide concentration in manure amended composting was observed throughout the trial with the highest concentration observed on the first day (120 ppm) and again an abrupt increase in concentration (70 ppm) was observed on the last day of composting. The author attributed high N_2O concentration to denitrification caused by high availability of nitrate ions to microorganisms. The author mentioned that nitrate is produced during biodegradation of green waste through nitrification.

Hellebrand (1998) also studied N_2O emissions and effect of ventilation rate at a specific temperature in a laboratory-scale set up. The apparatus was designed to fix ventilation rate between 10 and 1000 L min^{-1}. Gas emissions were analyzed at three different aeration rates; 20, 50, and 100 L $min^{-1}$and gas concentrations were analyzed by a spectrometer and a multi-gas monitor. The author found similar emission patterns for CH_4 and N_2O at all three aeration rates. The maximum CO_2 concentration was observed in the air leaving the substrate after Day 1. Nitrous oxide concentration was detected after the downfall in CO_2 peak. The N_2O concentration (200 ppm) was observed with a ventilation rate of 100 L min^{-1}. N_2O concentration increased to 4000 ppm at a ventilation rate of 20 L min^{-1}. The study concluded that increasing ventilation rate to100 L min^{-1} reduced the N_2O emission. The author concluded that N_2O emissions from composting can not be avoided but could be kept at minimum level with sufficient aeration.

Osada *et al.* (2000) carried out an experimental study to determine the effect of aeration rates on N_2O and CH_4 emissions from a swine waste composting. Swine waste (18 kg) was mixed with equal amount of sawdust to optimize moisture content (65%) for ideal composting conditions. Composting was done in a reactor (52 L volume) equipped with an aeration pump, an air flow meter, and a data logger. The authors found high emissions of N_2O and CH_4 gases during early stages of composting. The authors attributed high emissions during the early stages to the production of N_2O by denitrification of NO_x-N resulting from slightly anaerobic conditions. At aeration rate above 40 L m^{-3} min^{-1}, CH_4 emissions were remained less than 30 g m^{-3} during composting. Methane emissions were maximum at an aeration rate of 20 L m^{-3} min^{-1}. Similar N_2O emissions patterns were also observed during this experiment. Nitrous oxide emissions were observed decreasing with increasing aeration rate. The maximum N_2O emissions (72 g m^{-3} during composting) were measured at 20 L m^{-3} min^{-1} aeration rate.

Beck-Friis *et al.* (2000) conducted an experiment to analyze N_2O and CH_4 emissions from compost heaps of organic household wastes. The organic waste was mixed with chipped branches. The emissions were compared from four compost heaps 2-2.5 m high, 5-6 m wide and 15-20 m long. Another smaller pile of 1.2 m high, 2 m wide and 3 m long was formed. The small pile was turned once a week during the first two weeks and every fortnight thereafter using a bucket loader. Large heaps were turned at intervals of two to four weeks. Gas samples were taken at three different locations – in the centre of pile, along the slope and at the edge.

Beck-Friis *et al.* (2000) found a maximum N_2O concentration (106 $\mu L\ L^{-1}$) after 2 weeks of composting taken from the centre of a large heap at 2 m depth from surface. The concentration reduced to 43 $\mu L\ L^{-1}$ after 52 weeks of composting in large heap. Similar trends were observed in CH_4 analysis. Methane concentration measured at 2 m depth in the centre profile was reduced to 5 $mL\ L^{-1}$ from an initial value of 159 $mL\ L^{-1}$ observed during time span of 52 weeks. The authors found significant differences in concentration of N_2O and CH_4 gases in the large and small heaps. Nitrous oxide emissions from the surface of the large and small heaps increased with composting time. Nitrous oxide emissions in large heap reached at 480 $mg\ m^{-2}\ d^{-1}$ after 40 weeks of composting compared to 7 $mg\ m^{-2}\ d^{-1}$ measured after 2 weeks of composting. The authors predicted that bacteria present in compost could denitrify and some of them might survive the thermophilic phase. Maximum CH_4 emissions were measured from large compost heaps of 14 and 18 weeks old, ranging from 113-119 $g\ m^{-2}\ d^{-1}$. The authors indicated that higher CH_4 emissions at that interval may be due to low O_2 availability and prevailing anaerobic conditions resulting from compaction of material inside large heap. Again higher CH_4 and N_2O emissions were reported from large heaps compared to the small one.

Becks-Friis *et al.* (2001) also conducted a study on gaseous emissions of CO_2 and N_2O from organic household waste in a compost reactor under different controlled temperature conditions (O_2 concentration 16%, moisture content 65% w/w and 55°C temperature). A constant temperature (55°C) was obtained rapidly using external heating during the first trial whereas the natural self-heating process of biomass was used to raise temperature slowly to 55°C during second trial. The results suggested that patterns of CO_2 were significantly influenced by heating methods. The emissions rate of CO_2 was found to be low in the reactor with external heating. The authors found positive correlation between high rate of carbon decomposition and natural self-heating technique. In self-heating process, favorable conditions for microorganism resulted in high CO_2 emissions due to increase in microbial activity. No significant difference was found in

N_2O emissions from both types of heating systems. Nitrous oxide emissions were very low, 0.4% from natural self-heating reactor and 0.01% from external heating reactor.

Hao *et al.* (2001) analyzed GHG emissions from feedlot manure composting using two different composting methods: passive and active aeration. In passive aeration, the compost pile was never turned and air was supplied using open-ended perforated steel pipe. In active aeration, the compost pile was turned six times during the composting process. They studied the effect of turning and aeration method on GHG emissions.

Three replicated piles of both treatments were constructed using the same kind of cattle manure. GHGs were collected by placing a vented chamber on the surface of the compost piles. Gas samples were analyzed using gas chromatography (GC). Gas fluxes were calculated by plotting gas concentration against time using second-order polynomial equation ($C = a + bx + cx^2$), where C is the concentration of gases and x is the time (min). The emission rate for each half-hour was obtained taking derivative of equation at $t = 0$. The half-hour emissions rates were integrated over a 24-h period to obtain daily emissions rate by assuming that the half-hour flux represented the daily emission rate. The researchers found that, in passive aeration treatments daily emissions rates of CO_2, CH_4, and N_2O varied between 0 to 0.22 kg C m^{-2}, 0 to 0.4 kg C m^{-2}, 0 to 0.6 kg C m^{-2}, respectively. The GHG emissions were high during the first 30 days of composting but declined to zero thereafter. Total GHG emissions were 240.2 kg C per ton of initial dry manure. Whereas in active composting daily emissions rates of CO_2, CH_4, and N_2O varied between 0 to 0.35 kg C m^{-2}, 0 to 0.02 kg C m^{-2}, 0 to 0.92 kg C m^{-2}, respectively. The emissions from active composting were significantly higher than emissions from passive composting. Total GHG emissions in terms of CO_2 equivalents were 401.4 kg C per ton of initial dry manure from active composting. The authors attributed lower GHG emissions from passively aerated composting due to the incomplete decomposition and a lower gas diffusion rate. The lack of turning was found to be a main factor for reducing the rate of the decomposition process. In active aeration treatments, regular turning was introducing fresh air into the windrow and was a main factor for CO_2 production throughout the composting period. Their study found no significant difference in CH_4 emissions in active and passive composting although CH_4 emissions from active composting were slightly higher than from passive composting. Nitrous oxide emissions were very low compared to CO_2 and CH_4.

Hellebrand and Kalk (2001) analyzed CH_4 and N_2O emissions during windrow composting of cattle and pig manure. The windrows of animal waste (cattle and pig manure) mixed with straw bedding were formed and seven random points were selected on the windrows for gas sampling. The authors divided the gas emission from composting into four phases. Total gas flux (g m^{-2}) for CO_2, CH_4, and N_2O were presented for each phase. In each phase, CH_4 was generated only in first three weeks of composting. The authors attributed this phenomenon to evolution of anaerobic zones due to improper mixing of material in a pile. In anaerobic zones, CH_4 was generated by anaerobic bacteria until aerobic conditions prevailed in the piles. The mean CH_4 flux from the four phases throughout the trial was 1346 g m^{-2}. They concluded that complete anaerobic conditions due to compression of substrate can be avoided by adopting layer-system composting. Nitrous oxide emissions were high after three weeks of composting in each phase. The authors explained that nitrification and denitrification processes due to degradation of substrate were the main reason for N_2O production. The mean N_2O gas flux from four phases throughout the trial was 12.8 g m^{-2}.

Fukumoto *et al.* (2003) analyzed the effect of compost pile on N_2O and CH_4 emission patterns during swine manure composting. Fresh swine manure was mixed with sawdust to obtain initial moisture content 65%. Compost was piled up in conical shaped heap inside a room and was turned biweekly. The experiments were conducted simultaneously on small scale (Run 1) and larger scale (Run 2) compost piles. Small scale piles had 350 kg fresh mass, 0.7 m height and 1.4 m base diameter while large scale pile was having 780 kg fresh mass, 0.9 m height and 2 m base diameter. Fukumoto *et al.* (2003) found high CH_4 emissions (1.9 g kg-OM^{-1}) in the large scale pile (Run 2) compared to CH_4 emissions (1.0 g kg-OM^{-1}) in the small scale pile (Run 1). The authors attributed that high CH_4 emissions in Run 2 were due to increased number and size of anaerobic sites inside the bigger pile. They suggested that changing the scale of the compost pile was a main factor of difference in emission patterns. Similar patterns of N_2O emissions were observed with high N_2O emission (47g kg-TN^{-1}) from Run 2 compared to low N_2O

emissions (37.2 g kg-TN^{-1}) from Run 1. The authors correlated the low N_2O emissions from Run 1 with high aeration flow rate availability due to small-sized pile. The greater number of anaerobic sites in the large compost pile stimulated denitrification of nitrite at the centre of pile and consequently more and more N_2O was produced as a by-product. Their results indicated that N_2O emissions only occurred after the temperature and NH_3 concentration decreased in the compost pile.

Thompson *et al.* (2004) examined CH_4 and N_2O emissions from swine manure and wheat straw composting. The emissions were compared from two types of in-vessel composting, one with forced aeration and other without aeration. Composting was accomplished at two different places (Ridgetown and Chepstow, ON, Canada). Composting facilities were equipped with aeration fans installed under the composting floors. At the end of composting period, material was transferred to a curing pad in open space. The emissions were also measured using two chambers during the curing stage.

Thompson *et al.* (2004) found large variability between CH_4 and N_2O fluxes measured from the two composting sites. However, a similar pattern of low fluxes of CH_4 and N_2O during the first 13 days of in-vessel composting and a gradual increase in emissions afterwards were found in both the sites. The authors attributed increased N_2O emissions during in-vessel composting to the additional application of N rich manure into the piles. The increase in production of N_2O was observed after each manure application. The N_2O emissions during the curing stage varied significantly and were found to be higher than emissions measured during in-vessel composting from both the sites. The authors correlated this phenomenon with reduced concentration of available C during the curing stage. Less availability of C reduced immobilization of mineral N compounds, which increased the NH_4^+ content available for nitrification, and resulted in more N_2O production. Nitrous oxide emissions from the Chepstow site were found almost double compared to Ridgetown site during the curing stage. The authors explained this difference due to ideal temperature (mesophilic; 40°C) for N_2O production in the curing pile at the Chepstow site, whereas average temperature of the curing pile at the Ridgetown site was quite high (thermophilic; 58.6°C), which is considered non-favorable for N_2O production. The N_2O emissions during composting from Ridgetown and

Chepstow site were measured as 1849 and 1952 ng m^{-2} s^{-1}, respectively. Nitrous oxide emissions from the Ridgetown and the Chepstow site during curing stage were measured as 10960 and 20600 ng m^{-2} s^{-1}, respectively.

Thompson *et al.* (2004) found CH_4 emissions were found to gradually increase during in-vessel composting. The authors attributed the increasing pattern to the increased moisture and available C content together with manure addition during the initial stages. Methane emissions were increased two-fold towards the end of in-vessel composting in the Chepstow site while CH_4 emissions from the Ridgetown site were decreased at that time. The authors explained that aeration fans were only turned-on during 12 h of composting at the Chepstow site and this resulted in more CH_4 emissions due to anaerobic conditions, whereas aeration was provided at the Ridgetown site throughout the in-vessel composting which reduced the anoxic sites. Methane emissions from the Ridgetown and Chepstow sites during the curing stage were measured as 1031 and 729.5 μg m^{-2} s^{-1}, respectively. At the Chepstow site, CH_4 emissions during the curing stage were less than emissions during the in-vessel composting stage. Also, CH_4 emissions during the curing stage were low at the Chepstow site compared to emissions at the Ridgetown site. The authors attributed decreased emissions at the Chepstow site to increased circulation of air due to the small curing pile and low soluble C content. Higher CH_4 emissions at the Ridgetown site during curing were attributed to increased depth of the compost stacking.

Wolter *et al.* (2004) quantified GHG emissions from a pilot scale study on a conical pig manure pile. Gas emissions were collected using a tent around 4.4 m^3 of deep litter pig manure pile. The emissions were calculated by subtracting initial concentration from final concentration over 10 min intervals. The authors selected the 10 min interval due to low disturbance of mass transfer between the pile and ambient air during the short enclosure time. GHG emissions were also collected using a covered box from three different regions: top surface, upper half surface; and lower half surface of the pile. The authors concluded that emissions calculated from the covered box method were 3-7 times lower for CO_2 and N_2O than measured from the tent method due to high spatial variability of gas emissions through the surface of the pile. Due to the inhomogeneous structure of pile, the covered box might not detect all the gas emissions. The authors found most of the

GHGs were emitting from the top surface and upper half of the piles. Methane emissions were a maximum of 63 mg kg^{-1}on Day 27 and decreased to 6 mg kg^{-1}on Day 50. The authors attributed increased CH_4 emissions to anaerobic decomposition within the pile due to high thermophilic temperature. However, the authors could not find any correlation between N_2O concentration and temperature. Nitrous oxide emissions were relatively high at the beginning. The highest N_2O emissions were observed at Day 10. The results of N_2O emissions contradict the results of other studies (Hellmann *et al.* 1997). The authors attributed the differences in N_2O emission results to different origin of manure.

2.8.3 Anthropogenic greenhouse gas emissions

Carbon dioxide, methane and nitrous oxide are the primarily GHGs emitted from the combustion of fossil fuels. The CO_2 emissions from fuel combustion are primarily dependent on the carbon content of the fuel while CH_4 and N_2O emissions depend upon fuel type, vehicle technology, and pollution control devices (Environment Canada 2003).

The associated anthropogenic GHG emissions in manure management are from fuel combustion during: a) material acquisition and transport; b) establishment of the piles; c) turning of the pile; and d) spreading of residuals.

GHG emission factors for fuel consumption

The emission factors used here to estimate GHGs emissions from diesel fuel combustion were developed by Environment Canada in consultation with various international organizations, government departments, and industry associations (Environment Canada 2002). The emissions factors were developed from information obtained from scientific papers, engineering calculations, published reports, and engineering judgment. The GHG emissions were calculated by multiplying a base quantity of consumed fuel by an emission factor appropriate to the type of fuel and the particular source and type of technology. The GHG emission rate from a specific fossil fuel can be estimated using the following equation:

$$E = BQ \times EF \quad \ldots\ldots\ldots\ldots\ldots\ldots 2.1$$

Where,

E = Emission rate (g-GHG)

BQ = Amount of fuel burned for particular vehicle

EF = Emission factor (g-GHG/amount of fuel burned).

The emission factors used in this study to convert amount of fuel consumed into GHG emissions are listed in Appendix 1.

2.9 SUMMARY OF LITERATURE REVIEW

This chapter provides current available information and baseline data on GHG emissions from beef feedlot composting. A detailed literature review on GHG emissions suggested that there are still uncertainties and variations involved in GHG estimations, mainly due to a lack of standard methodology and the use of different types of feedlot manure. There is a limited number of studies which quantify biogenic and non biogenic GHG emissions from the manure and compost pile using beef feedlot manure. Some researchers have found low GHG emissions from unturned piles due to incomplete decomposition, as compared to a turned windrow (Hao *et al.* 2001). On the other hand, some studies suggested that large unturned pile can emit high methane due to the development of anaerobic microsites in the pile (Fukumoto *et al.* 2003). The limited number of studies related to GHG emissions from beef feedlot composting calls for further research to quantify GHG emissions.

Chapter 3. Materials and methods

3.1 SITE DESCRIPTION OF BEEF FEEDLOT AND FACILITY OPERATIONS

The field study was conducted at the Agriculture and Agri-Food Canada Research Centre at Lethbridge, Alberta (AAFC Lethbridge) in collaboration with Alberta Agriculture, Food, and Rural Development (AAFRD). The project started on July 14, 2004 and finished on October 28, 2004. On July 14, a beef cattle feedlot at AAFC Lethbridge was cleaned out using a front-end loader. The manure pack contained cattle manure and wheat straw as bedding material, which had been spread on the feedlot before the arrival of the animals and added as required during the animal occupancy. The ratio of manure to bedding material was about 4:1 on a mass basis.

The material was transported using a dump truck of 5000 kg carrying capacity (S1700, International Truck and Engine Corp., Warnerville, IL) to the selected storage site 200 m distance from the feedlot. Two different piles were constructed from the same feedlot manure parallel to each other in an east-west orientation (Figure 3.1). The initial mass of manure for the compost windrow and the manure stockpile was determined by weighing each truckload using an electronic scale (Model-8142, Mettler Toledo, Mississauga, ON). The initial mass in each pile was recorded and approximately the same mass of material was included in each. The compost windrow was formed with a parabolic cross-section (45 x 2.5 x 1.1 m) using a skid-steer loader (135-S, Thomas Equipment Ltd., Mississauga, ON). The manure stockpile was formed in the shape of a trapezoidal heap (21 x 3.6 x 1.52 m).

Figure 3.1 Compost pile and manure stockpile establishment

3.2 SYSTEM BOUNDARIES

For the purpose of greenhouse gas (GHG) emission analysis in this study, the system boundaries were defined. The GHG emissions arising from decomposition of manure before pen cleaning were not included within the system boundaries. The following sources of GHG emissions were included in the estimation of the total emissions from the two manure handling systems: (1) Non-biogenic CO_2, CH_4, and N_2O emissions related to the combustion of fossil fuel in material acquisition and transportation using a front-end loader and dump truck; (2) Biogenic CO_2, CH_4, and N_2O emissions from natural decomposition of organic material; (3) CO_2, CH_4, and N_2O emissions related to the combustion of fossil fuel by a windrow turner during composting; and (4) CO_2, CH_4, and N_2O emissions related to the combustion of fossil fuel by a tractor-driven manure spreader while spreading the end-product on the field. The long-term GHG emissions after field application of the end-products of the compost pile and manure stockpile were not included within the system boundaries.

3.3 MOISTURE CONTENT

Manure samples from the compost windrow and the manure stockpile were collected at regular intervals throughout the trial to determine the moisture content present in the piles. Samples were collected from four selected locations in each pile: a central location at each side of the pile and one location 5 m from each end of the pile. At each location, material was sampled from two different depths: 0.3 m and 0.9 m. The reason for collecting the samples at different depths was that moisture content often varies with depth (Rynk 1991). Approximately 2 kg of manure was collected from each of the four locations using a shovel. All the samples from the four locations were mixed thoroughly in a container to homogenize the manure from each pile. About 1 kg of each mixture was sealed in a resealable plastic bag until analysis. The moisture content of each sample was determined by oven-drying five 100 g sub-samples of manure at 65°C for 24 hours in aluminum trays. The difference in the mass of the manure before and after oven drying was measured using an electronic balance (AC-12K, Denver Instruments, Denver, CO.). The moisture content of the each sub-sample manure was then calculated using the following equation:

$$\text{Moisture content (\% wet basis)} = \frac{\text{Wet mass - Dry mass}}{\text{Wet mass}} \times 100$$

The moisture content of the overall sample was calculated by taking the average of the five sub-samples.

3.4 BULK DENSITY

The initial and final bulk density of the wet material was measured using an aluminum pail of known volume (0.07 m^3). The weight of the empty and filled pail was measured using an electronic balance (Sartorius EB60FEG-L, Precision Weighing Balances, Bradford, MA). Material was loaded into the pail manually using a shovel. The pail was dropped on a firm surface three times from a height of approximately 150 mm and topped off with manure before weighing. Two duplicate readings were taken from the manure stockpile and the compost pile.

3.5 NUTRIENT CHARACTERIZATION

The initial and final manure samples were tested for C:N ratio, total C, and total N. The same collecting method as mentioned above in moisture content analysis was used to collect the initial and final samples. Approximately 1 kg of sample from the manure stockpile and 1 kg from the compost pile were collected in resealable plastic bags. The samples were preserved in a cooler at $4^{o}C$ until analysis. The C:N ratio analysis was performed at Enviro-Test Laboratories (Saskatoon, SK). To verify the results of the analysis conducted by Enviro-Test Laboratories, analysis was also performed at AAFC Lethbridge. Total carbon was calculated using total combustion method and total nitrogen was calculated as Total Kjeldahl Nitrogen using titration method.

3.6 TEMPERATURE AND WEATHER MONITORING

The internal temperature of a compost pile or manure stockpile is usually considered as a good indicator of biological activity (Rynk 1992). Temperature was monitored on gas sampling days throughout the course of the study. The temperature of the compost pile was recorded before and after turning of the compost. Temperature was also monitored approximately 30 minutes before the start and 30 minutes after the finish of gas sampling. Temperature was measured at 10 equally spaced locations on each side of the compost and manure piles. A dial-gauge thermometer with a 0.9 m-long stem (Model-20, Trend Instruments Inc., Houston, TX) was used to monitor the temperature (Figure 3.2). Temperature measurements were made at 0.33 m and 0.90 m depths at each location. The temperature probe was inserted into the pile and observed until the temperature reading stopped changing (approximately 3-4 minutes after insertion).

Weather data (mean hourly ambient temperature, wind velocity, and rainfall) were obtained from a weather station located on the periphery of AAFC Lethbridge, 500 m from the experiment site.

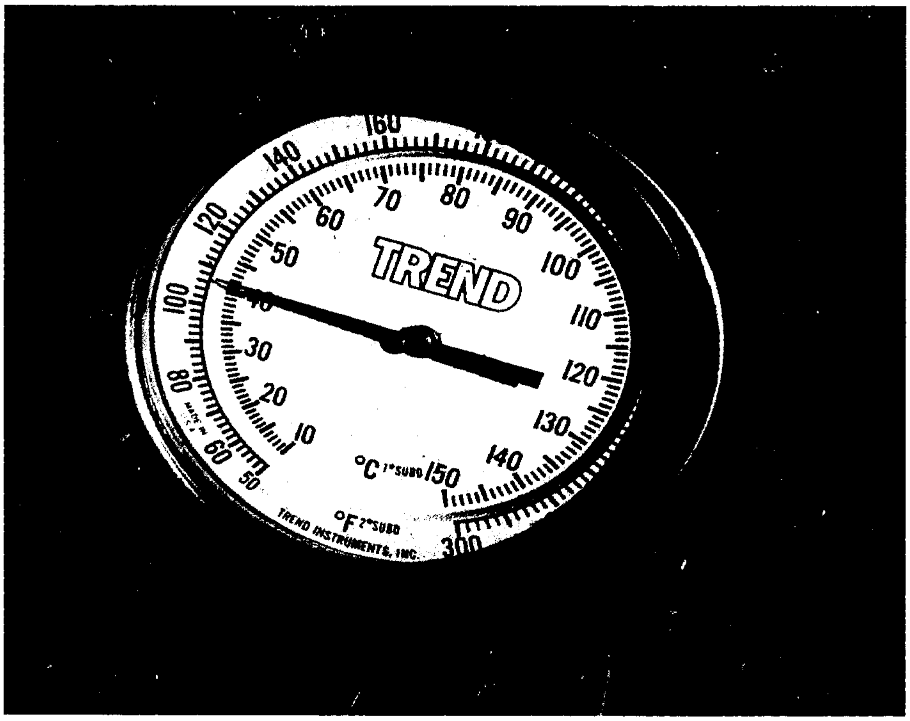


Figure 3.2 Dial-gauge thermometer

3.7 TURNING

A tractor-driven, rotating drum-type windrow turner (Earthsaver CT-12, Fuel Harvesters Corp., Midland, TX) was used to turn the compost windrow. The rotating drum (3.6 m long) of the turner had a diameter of 0.28 m and 112 blades of 200 mm length. A hydraulic mechanism was used to adjust the vertical height of the drum from 0.20 m to 0.46 m. The rotating drum turned at approximately 600 revolutions per minute. A tractor (8360, Ford New Holland, New Holland, PA) with 83.8 kW (115 hp) was used to pull the turner (Figure 3.3). The turner was powered from the tractor power-take-off shaft. The forward speed of tractor was kept around 5 km h^{-1}.

Figure 3.3 Windrow turning operation using tractor-operated turner

The compost windrow was turned once a week for the first four weeks of the trial and then once every two weeks. The compost was therefore turned 8 times (Day 8, Day 15, Day 22, Day 29, Day 36, Day 43, Day 57, Day 79) throughout the trial, with the first turning occurring on July 22, 2004. The compost windrow was turned alternatively from either end. The time required to turn the compost windrow during each turning was observed and the length of the windrow was measured after each turning. The compost windrow was reshaped using a skid-steer loader (135-S, Thomas Equipment Ltd., Mississauga, ON).on Day 36. At that time, the length of the compost windrow was reduced (22.4 m) whereas the width (3.5 m) and height (1.7 m) were increased.

3.8 WATERING SYSTEM

As described, the moisture content of the manure was measured regularly. Due to the turning operation and the warm and windy climate, more moisture was lost from the compost pile than from the manure stockpile. To bring the moisture content to the range of 50%-55% from the observed moisture content (34%) on Day 36, approximately 6000 L of water were added to the compost windrow after turning. A watering system capable of spraying 8 L s^{-1} was used to spray the top and sides of the compost windrow. The truck-mounted spraying system was driven along both sides of the windrow.

3.9 VOLUME REDUCTION

The total volume of material in the compost pile and the manure stockpile was estimated using a windrow profiler (Figure 3.4).

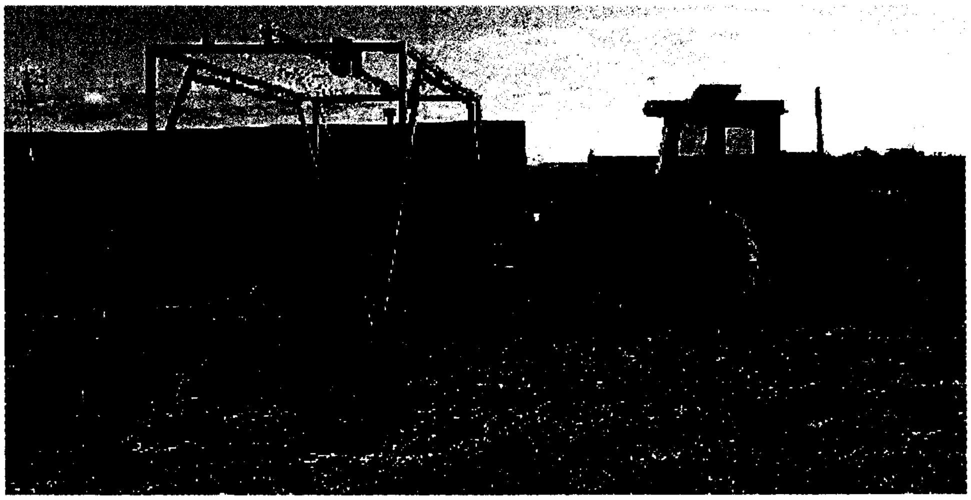


Figure 3.4 Windrow profiler

The windrow profiler was manufactured at the AAFRD Agricultural Technology Centre (Lethbridge, AB). The windrow profiler scanned the cross-section of a pile with an overhead-mounted laser range finder (NR-40, Nova Ranger, San Diego, CA) which moved horizontally on a frame over the top of the pile. The range finder determined the distance from the frame to the top of the pile as it moved horizontally from one end of the frame to the other. The laser measured the cross-section of the pile by taking 216 height measurements while moving 4.41 m laterally. An onboard data-acquisition system transferred the information to a laptop computer installed in the tractor. The data were later analyzed using Euler's trapezoidal integration method (Davis and Robinowitz 1984). An example of the Euler integration method is shown in Figure 3.5.

The shaded area in Figure 3.5 is used as an approximation of the area under the curve H(x) between two points [X_1, H_1] and [X_2, H_2].

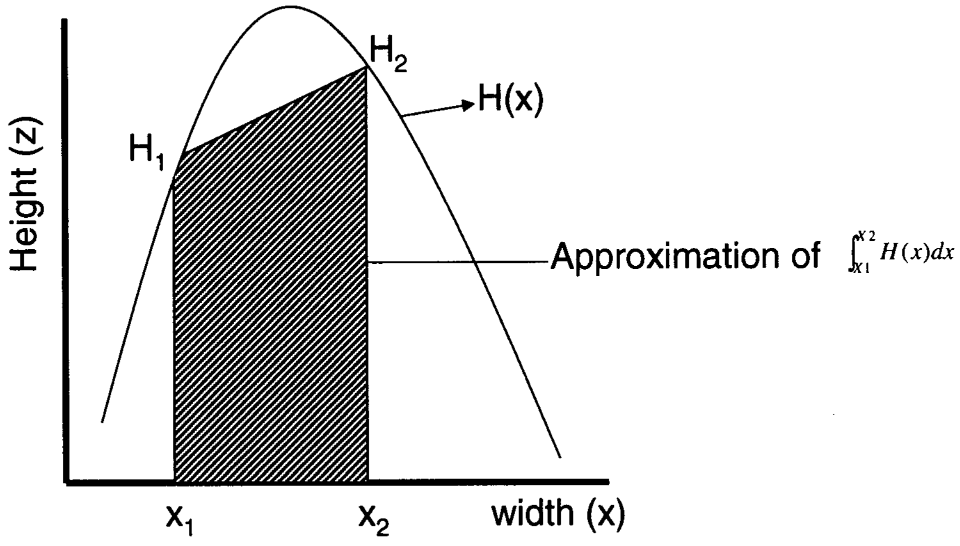


Figure 3.5 Trapezoidal approximation of area

$$A_j = \int_{x=0}^{W} H(x)dx \cong \sum_{i=1}^{m-1} \frac{\Delta x}{2}\left[H_i + H_{i+1}\right] \ldots\ldots\ldots\ldots\ldots\ldots 3.1$$

At each observation point, the volume of material inside each pile was calculated by multiplying the average cross-sectional area (A_j) by the incremental length (Δy) of the windrow:

$$V = \int_{y=0}^{L} \int_{x=0}^{W} H(x,y)dxdy = \sum_{j=1}^{n-1} \frac{\Delta y}{2}\left[A_j + A_{j+1}\right] \ldots\ldots 3.2$$

Where,

x= 0 to W indicate the total width of the pile, (m)

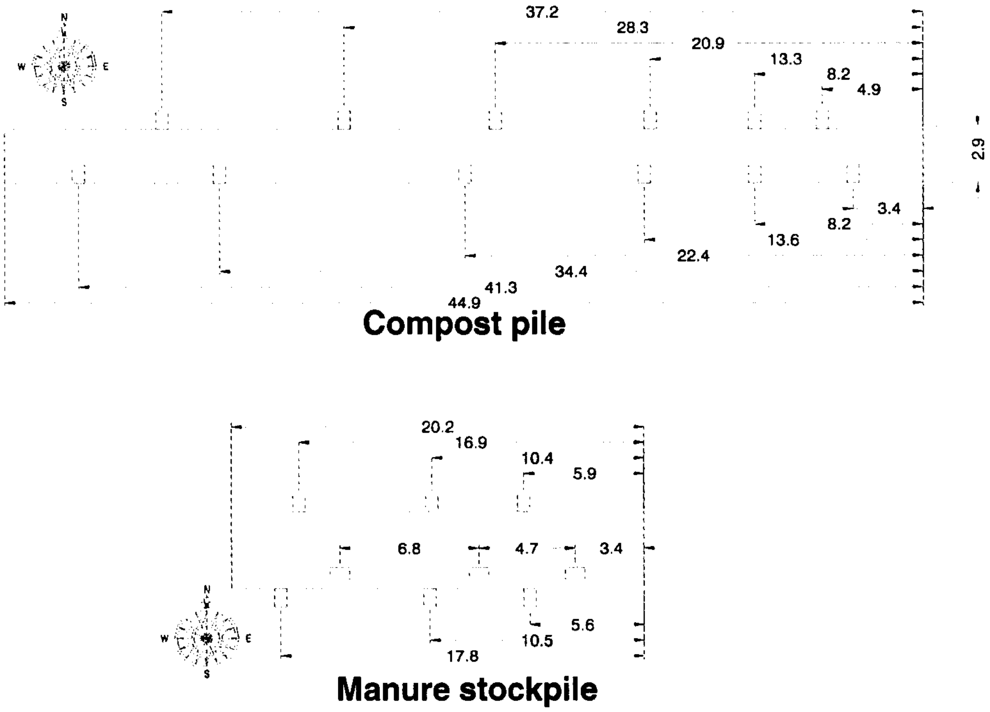
Δx= Incremental width, (m)

y=0 to L, Length of pile, (m)

i= Index of measurements in a cross-section (x). i = (1, 2, 3,....., m); m = 216

j= Index of cross sections along the length, j = (1, 2, 3,....., n); n = 9 for manure or 12 for compost

The windrow profiling was done four times throughout the course of the composting period: during the first week of the trial (Day 6) before and after the reshaping of the compost windrow (Day 43), and during the last week of the project (Day 85). The windrow profiler was run at 12 and 9 different locations over the compost windrow and the manure stockpile, respectively (Figure 3.6).



Compost pile

Manure stockpile

Figure 3.6 Windrow profiler scanning locations. (Dimensions are in meters.)

3.10 BIOGENIC GREENHOUSE GAS COLLECTION

Biogenic GHGs were collected from the surface of the compost and manure piles using a static chamber technique (Figure 3.7). The technique has been used in several similar studies (Hao *et al.* 2001; Czepiel *et al.* 1996; Hellmann 1997; and Teshima 2001). The metal, cylindrical chamber had an open base, a volume of 3.33 L, a height of 180 mm, and a base area of 0.0184 m^2. After Day 15, the static chambers were covered with a plastic container while sampling to shield them from sun and wind. Off-gas samples were collected from the top of the chamber through an opening fitted with a butyl septum. Gas samples were drawn through the butyl septum with a 30-mL gas syringe equipped with a 20-gauge needle having outer and inner diameters of 0.904 and 0.584 mm, respectively.

Sampling was performed in each of equal length along the length of pile into three equal sections. A static chamber was placed in each zone. Two duplicate samples were collected at 0, 5, 10, 20, and 30 min from each chamber on every sampling day. A total of 600 biogenic GHG samples were thereby collected on Days 1, 2, 8, 15, 22, 29, 36, 43, 57 and 79 from the compost and manure stockpile.

Figure 3.7 Static chamber for compost off-gas collection

Each gas sample was transferred into a 10-mL evacuated vial (Exetainer™, Labco Limited, Buckinghamshire, UK). The vials were pre-evacuated to 90 kPa using a vacuum pump (SpeediVac, Sussex, UK) attached to a custom-built manifold of four hypodermic needles. The sample vials were transported in an ice-packed walk-in cooler where they were stored at 4°C until analysis.

3.11 BIOGENIC GREENHOUSE GAS ANALYSIS

Analysis of GHGs was done at a laboratory in the Department of Agricultural, Food and Nutritional Science (AFNS), University of Alberta (Edmonton, AB). Greenhouse gas concentration was analyzed using a gas chromatograph (GC). Carbon dioxide and methane gas concentrations were analyzed using a Hewlett Packard 5890 Series II GC (Agilent Tech. Co., Mississauga, ON) equipped with a thermal conductivity detector. The gases were separated on a HP-Plot Q capillary column (30 m x 0.053 mm ID) (Agilent Tech. Co., Mississauga, ON). The injector and column temperatures were maintained at 50°C and the detector temperature was maintained at 80°C. The column head pressure was kept at 27.5 kPa and the injector was set for 20:1 split injection. Data integration was performed using Shimadzu Class VP Software (Chromatograph Laboratory Automated Software System Version 4.2, Shimadzu Scientific Instrument Inc., Columbia, MD).

Nitrous oxide gas concentrations were analyzed using a Varian Start 3400 Series GC (Analytical Instruments Inc., Golden, CO) equipped with an electron capture detector. Nitrous oxide gas was separated on a Poropak-Q S 80/100 stainless steel capillary column (1525 x 3.2 mm ID) (Analytical Instruments Inc., Golden, CO). The injector and column temperatures were maintained at 60°C and the detector temperature was maintained at 300°C. Gas samples were injected automatically using a multi-port injector system. The column head pressure was kept at 206.85 kPa and the carrier flow rate was maintained at 30 mL min^{-1}. Varian G Star Chromatography Workstation Version-5 software (Analytical Instruments Inc., Golden, CO) was used to analyze the data.

Greenhouse gas concentrations were recorded in $\mu L\ L^{-1}$. The concentrations from the paired samples were averaged. According to the Fick's law, diffusion of gases from higher to lower concentrations is proportional to the concentration gradient (Nay *et al.* 1994). The concentrations were not increasing linearly over time due to the GHG concentration differences in a static chamber and pile. To accommodate the non-linear increase of concentrations, three different models were compared to estimate the best-fit curve to calculate the gas flux ($\mu L\ L^{-1}\ s^{-1}$): a linear regression model (Teshima 2003), a second order polynomial (Hao *et al.* 2004) and a logarithmic model (Hutchison and Mosier 1981).

- **Linear regression model**

A linear mathematical model, as results from linear regression analysis, is of the form:

$$Y = mX + C \quad \text{..............................} 3.3$$

Several researchers have used linear regression models to estimate best-fit curve to calculate the gas flux (μL L^{-1} s^{-1}) by assuming linear increase of gas concentration over time (Teshima 2003; Hellebrand 1998). The slope (*m*) of the linear equation 3.3 gives the concentration rates. An example of a GHG accumulation curve is show in Figure 3.8.

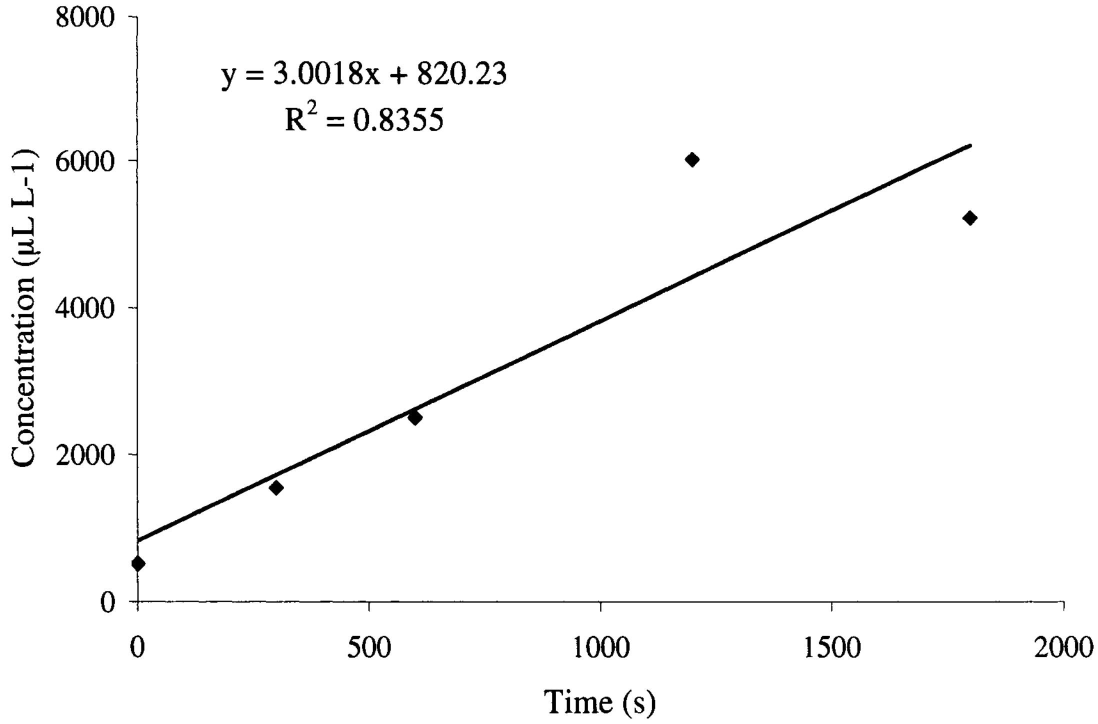


Figure 3.8 Greenhouse gas accumulation curve using linear regression model

- **Second order polynomial analysis**

Following the procedure described by Hao *et al.* (2004), a second order polynomial equation was tried as follow:

$$C = a + bx + cx^2 \quad \text{...} 3.4$$

Where C is the concentration of gases (μL L^{-1}) and x is the time (s) for each sampling time. The flux was calculated using the first derivatives of the second order polynomial at time (x) = $0\left(dC/dx_{x\to 0} = b\right)$. An example of a GHG accumulation curve is show in Figure 3.9.

The advantage of using a second order polynomial is that the assumption of linear increase of gas concentration over time within a chamber can be avoided. Another advantage of using nonlinear regression is that it can produce good estimates of the parameters in the model with relatively few data. Greenhouse gas concentration increases quickly at first and then levels off over time as discussed earlier. Linear regression models cannot describe such asymptotic dynamics. The disadvantage of using a second order polynomial is that this model is not asymptotic due to their convex or concave nature of curves. A second order polynomial is not a good model for the concentration change over the long period of time.

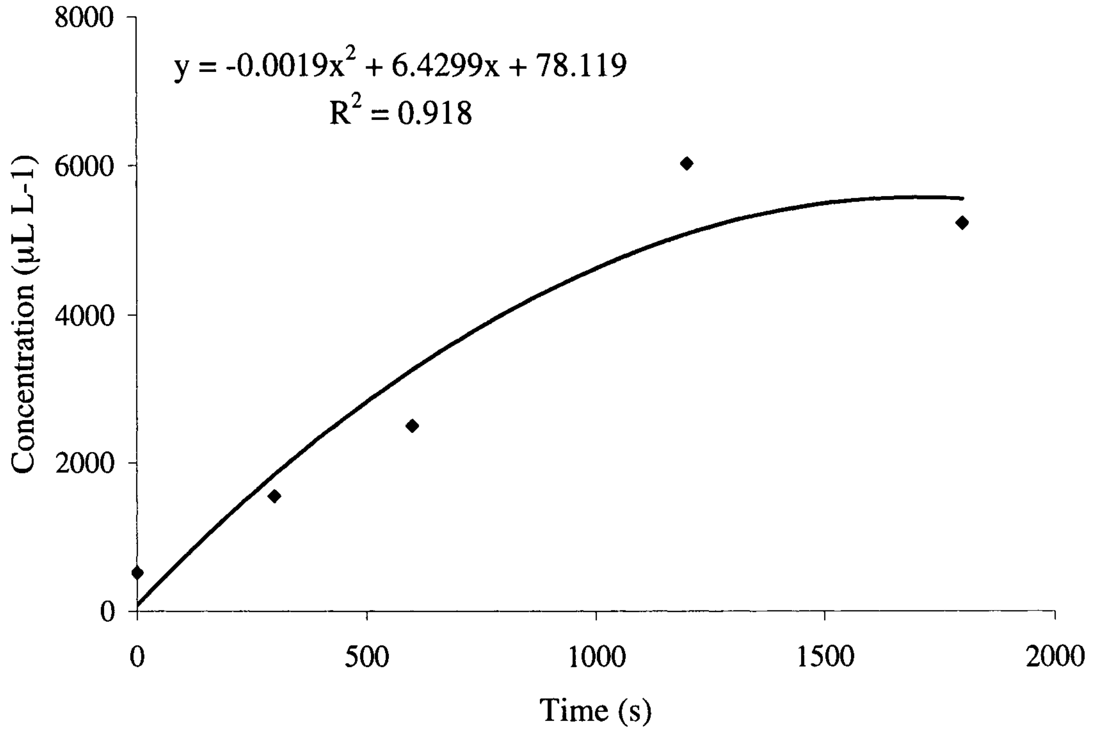


Figure 3.9 Greenhouse gas accumulation curve using second order polynomial

- **Logarithmic model (Hutchison and Mosier 1981)**

Hutchison and Mosier (1981) proposed an equation to estimate the gas flux inside the chamber as:

$$\dot{C} = \frac{1}{t} \times (c_1 - c_0)^2 / (2c_1 - c_0 - c_2) \times \ln((c_1 - c_0)/(c_2 - c_1)) \ldots\ldots\ldots\ldots\ldots\ldots\ldots 3.5$$

Where,

$\dot{C} =$ Rate of change in concentration (μL L^{-1} s^{-1}).

c_0, c_1, $and\ c_2$ = The concentration of gas at time 0, t, and 2t after placing the chamber on the emitting surface.

The Hutchison and Mosier (1981) equation overcomes the potential of errors associated with the second order polynomial and decrease in predicted gas concentration with time as the gas accumulates within the chamber. The proposed Equation 3.5 can describe the asymptotic phenomenon of gas accumulation in a closed chamber due to a steadily decreasing gradient between the partial pressure of gas in the soil and in the chamber. This approach provides a better representation, therefore, of the expected trend in gas concentrations in a closed sample chamber. Given the time-spacing of the measured concentration data, it is possible to calculate mass flux rate with Hutchison and Mosier model using either of two equal sets of time intervals (0, 300, and 600 s) and (0, 600, and 1200 s) using the Hutchison and Mosier model. For the purpose of discussion, further data analysis are performed using concentration values at (0, 300, and 600 s) time intervals.

The rate of change of the GHG concentrations were converted into emission rates using an equation based on the Ideal Gas Law (Equation 3.6). The Ideal Gas Law states that:

$$PV = nRT_k \ldots\ldots\ldots\ldots 3.6$$

Where:

P = Pressure (1 atm or 101.3 kPa at standard pressure)

V = Volume of gas (L)

n = Number of moles

R = Universal gas constant (0.08207 L atm mol^{-1} K^{-1})

T_k = Temperature (absolute, 273.15 K at standard temperature)

Rearranging Equation 3.6 yields an expression for the molar volume (v_m) of an ideal gas:

$$v_m = \frac{V}{n} \equiv \frac{RT_k}{P} \text{............} 3.7$$

Substitution of the values for standard temperature and pressure yields:

$$v_m = 22.4\ Lmol^{-1}$$

The rate of change of the concentration was converted into an emission rate (chamber mass flux) using the following equation:

$$\dot{m} = \left[\frac{\dot{C}}{10^6}\right]\frac{MV_s}{v_m} \text{.....................} 3.8$$

Where,

$\dot{m}$ = Chamber mass flux (g s^{-1})

$\dot{C}$ = Rate of change in concentration (μL L^{-1} s^{-1})

M= Molar mass (CO_2 = 44, CH_4 = 16, and NO_2 = 46 g mol^{-1})

V_s = volume of static chamber (3.33 L)

Dividing Equation 3.8 by the basal area of the static chamber ($A_s = 0.0184\ m^2$) yields mass flux per unit area ($g\ s^{-1}\ m^{-2}$).

$$\dot{f} = \frac{\dot{m}}{A_s} \qquad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 3.9$$

The mass flux per unit area ($\dot{f}$) was extrapolated over the entire pile by multiplying by the surface area of the pile. The piles were scanned at different locations using a windrow profiler in order to estimate cross-section perimeter. The surface areas of the piles were calculated by multiplying averaged cross-section perimeter of the piles drawn between two adjacent locations with the length of the piles between two locations. The surface areas at the first and last location of the piles were estimated by multiplying cross-section of the piles at the first and last location with the incremental length of the pile. The total surface areas of the piles were estimated by addition of each surface area at different locations.

Off-gases were only collected from the top of the pile. Previous studies indicated that the maximum emissions occur at the top of the pile (Fernandes *et al.* 1994; Wolter *et al.* 2004). There is always uncertainty involved in extrapolating emissions from a small area over the entire pile due to the high spatial variability of gas emissions through the surface of the pile. This phenomenon can be explained due to the high temperature in the centre of the pile and a chimney effect by which air is drawn through the lower parts into the pile and warm air moves upwards in the centre, resulting in convective aeration (Figure 3.10). The extrapolation of the emission rate over the entire pile by assuming uniform emissions throughout the pile may result in overestimation of emissions. Therefore, an assumption has been made at the end of study that only ½ of the total surface area of the piles are emitting off-gases. The comparison of GHG emissions between two piles are made on the 50% emitting area.

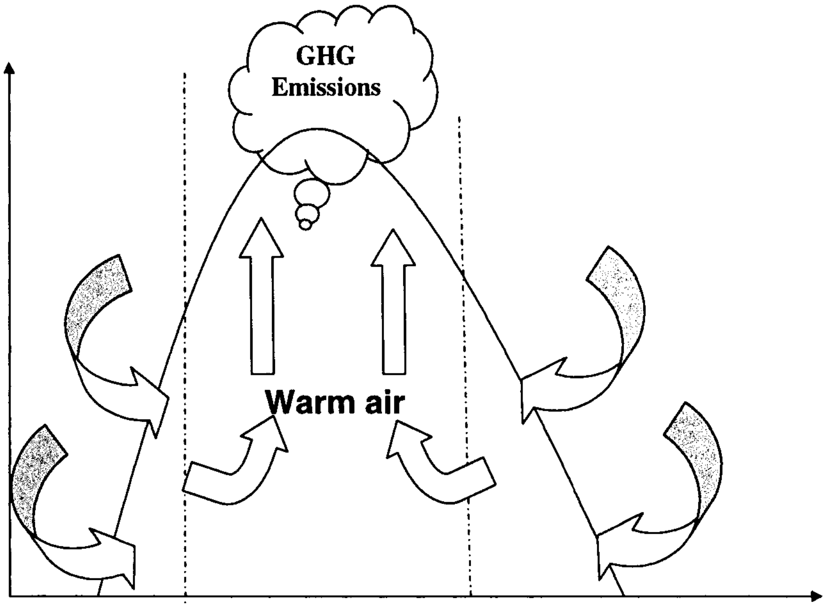


Figure 3.10 Movement of warm air and gases inside the pile

Greenhouse gas emissions obtained from the half-hour sampling time intervals were assumed to be representative of daily emissions, and the daily emission rate was, therefore, estimated by multiplying the half-hour emission rates by the 24 h interval. There is always uncertainty involved when extrapolating data over extended periods because emission rates vary with time depending on environmental conditions and other factors (Hao *et al.* 2001). Therefore, extrapolation of the instantaneous emission rates obtained from the half-hour time interval over the extended time might also have resulted in errors in estimation of emissions. Biogenic GHG emissions during the active composting phase from the compost pile and the manure stockpile were also expressed as mass CO_2-equivalent per unit time per unit initial dry mass by dividing the total pile emission rate by the initial dry mass.

3.12 SPREADING OF END PRODUCT

The end products from the compost pile and manure stockpile were spread on adjoining fields. A rectangular field 1.3 km from the site storage was selected to receive the end products. Diesel fuel consumption during spreading was measured. A tractor drawn, PTO-powered rotating-tyne manure spreader (Series II-226, Hydra-Spread, Windsor, ON) was used to spread the end products. The forward speed of the manure spreader was kept between 7.5-8 km h^{-1} to achieve an application rate of 50 t ha^{-1}.

3.13 NON-BIOGENIC GREENHOUSE GAS ANALYSIS

Machinery involved in material acquisition, transportation, turning, and spreading operations consumed fuel and contributed to GHG emissions. There were also other non-biogenic GHG emissions from other energy sources: the watering system used for spraying water on the compost pile and the tractor-mounted windrow profiler were not considered in this study.

Emissions estimates for CO_2, CH_4, and N_2O from fuel combustion were calculated by multiplying standard emission factors (Environment Canada 2002) with the amount of fossil fuel consumed by the machinery. Table 3.1 shows various equipment and their specification used in this study.

Table 3.1 Machinery involved in each operation

Operation	Machinery
Feedlot cleaning	WA120, Komastu America Corp., Vernon Hills, IL-Front-end Loader
Transportation of raw manure and establishment of piles	Dump Truck-S1700, International Truck and Engine Corp and Skid-steer Loader-135-S, Thomas Equipment Ltd., Bradford, ON
Turning of compost pile	Tractor-drivenWindrow Turner-Earthsaver CT-12, Energy Harvesters Corp., Midland TX
Transportation and application of end-product	Manure Spreader-Series II-226, Hydra-Spread, Windsor, ON

Raw feedlot manure was loaded onto a dump truck using a front-end loader and hauled with the truck to establish the compost pile and manure stockpile at the experiment site. The compost pile was turned using a tractor-driven windrow turner during the composting period. At the end of the study, the finished product was loaded onto a manure spreader with a Komatsu front-end loader and hauled to the application site. The amount of diesel fuel consumed by the machinery was measured during each operation. At the start of each operation, the fuel tank of each machine was completely filled at the AAFC gas station near the composting site and the initial level of diesel fuel in the fuel tank was marked. After each operation was performed, the machinery was driven back to the AAFC fill station. The fuel tank was topped up to the initial mark using a fuel pump and a 2-L graduated cylinder. The amount of diesel fuel required to fill the fuel tank was measured using the graduated cylinder.

Greenhouse gas emissions were estimated by multiplying emission factors (Appendix 1) (kg CO_2-equivalent L^{-1} of diesel fuel consumed) by the amount of fuel used in each operation (Environment Canada 2002).

3.14 STATISTICAL ANALYSIS

The measurements of the biogenic GHG emissions from the compost pile and the manure stockpile were analyzed with the SAS Mixed Procedure (SAS 2001). The residuals were tested for normal distribution using proc univariate. A mixed model with a repeated measures statement was used to test for significant effects. Each pile was divided into three zones and zones were assumed as replicates (pseudo-replicates). The before and after reshaping of the piles were treated as blocks in time. Treatments included in the mixed model (compost pile and manure stockpile) were treated as fixed factors. All other factors (blocks, time and zones) were random variables. Effects were deemed to be significant at $P < 0.05$. The statistical model used for the analysis was:

$$Y_{ijk} = \mu + Trt_i + Day_j + Block_k + Trt_i \mid Day_j \mid Block_k + e_{ijkl}$$

Where,

Y_{ijk}= GHG concentration

μ= Overall mean

Trt_i= Effect of i^{th} treatment (i = compost, manure)

Day_j= Effect of j^{th} day (j = 1, 2, 8, 15...79)

$Block_k$= Effect of k^{th} block (k= 1 i.e. before reshaping and 2 after reshaping)

$Trt_i * Day_j$ = Effect of interaction between the i^{th} treatment and j^{th} day

$Trt_i * Block_k$= Effect of interaction between the i^{th} treatment and k^{th} block

$Day_j * Block_k$= Effect of interaction between the j^{th} day and k^{th} block

$Trt_i * Day_j * Block_k$= Effect of interaction among the i^{th} treatment, j^{th} day and k^{th} block

e_{ijkl} = residual error.

The complete SAS code used to analyze the data is shown in Appendix 2.

Chapter 4. Results and discussion

This chapter includes a discussion of weather conditions during the trial, the initial and final properties of manure in the compost pile and the manure stockpile, biogenic GHG emissions resulting from decomposition, and non-biogenic GHG emissions resulting from combustion of fuel during associated activities.

4.1 WEATHER DATA

Table 4.1 Weather data during trial.

Sampling day	Average temperature (^{o}C)	Average wind velocity ($km\ h^{-1}$)
July 14, 2004	23.2	14.4
July 15, 2004	23.4	14.8
July 22, 2004	21.2	13.5
July 29, 2004	21.1	11.4
August 4, 2004	23.9	14.2
August 11, 2004	20.6	10.8
August 18, 2004	19.5	9.4
August 24, 2004	18.3	20.5
September 8, 2004	19.8	20.1
September 23, 2004	12.2	13.9
October 28, 2004	3.3	9.1

The weather data indicate that the average ambient temperature was higher during the initial stage of the composting period. The maximum average daily temperature (23.4^{o}C) was recorded on July 15, 2004 while the minimum average temperature (3.3^{o}C) was observed on October 28, 2004. The average daily wind velocity on sampling days was in the range of 9.1 to 20.5 km h^{-1} throughout the composting period.

4.2 MOISTURE CONTENT MEASUREMENTS

Proper moisture content (40% to 65%) is an important physical parameter to maintain optimum conditions for composting (Rynk 1992). Figure 4.1 shows the moisture content (M.C.) measurements from the compost pile and the manure stockpile. The initial M.C. in the compost and manure pile was 63% and 61%, respectively. It was observed that the M.C. of the compost pile decreased more rapidly than that of the manure stockpile. The M.C. on Day 36 was 36% and 58% in the compost pile and the manure stockpile, respectively. The reason for higher moisture loss from the compost pile was due to regular turning, whereas the manure stockpile was undisturbed and the M.C. remained above 50% throughout the trial (Figure 4.1). It was noticed that the manure stockpile developed a thick crust, which might have reduced moisture loss.

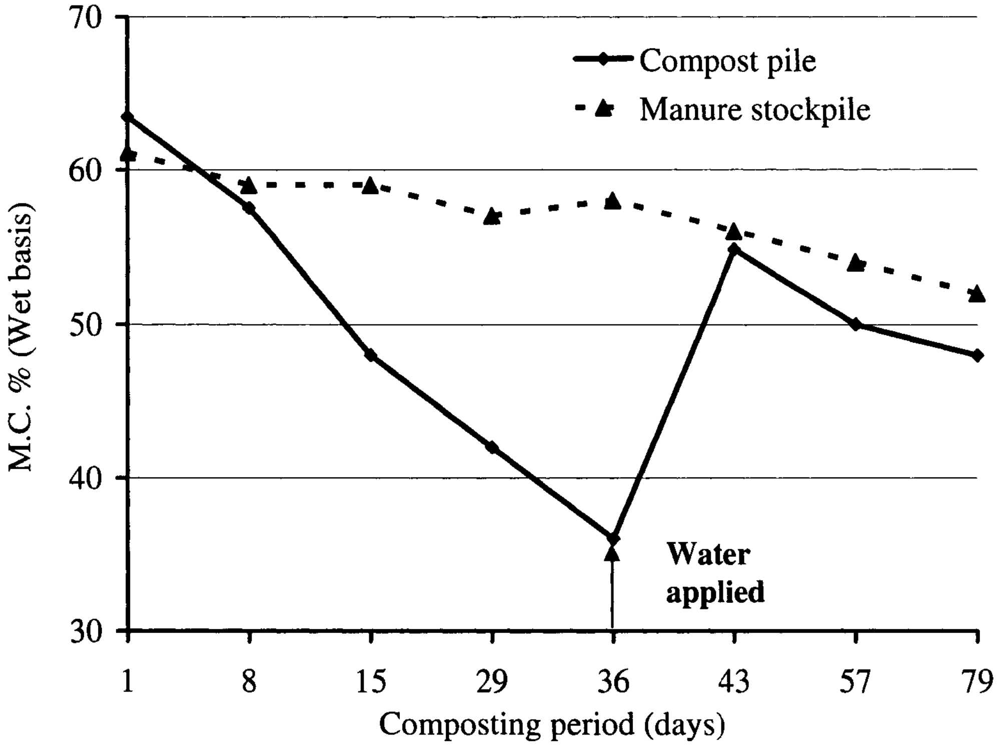


Figure 4.1 Moisture content of the compost pile and manure stockpile

The turning of the compost pile is believed to have increased the microbial activity by improving aeration, removing trapped heat, and exposing new substrate to the microorganisms in the pile (Rynk 1992). More aerobic conditions could have stimulated the microbial decomposition of organic matter and quickly released heat and water vapor during this process. The moisture content of the compost pile on Day 36 was 36%, which is less than the recommended M.C. range (40% to 65%) (Rynk 1992). Therefore, in order to increase the M.C., an additional 6000 L of water was sprayed onto the compost pile. The addition of water increased the M.C. to 55%, as measured the following week (Figure 4.1).

4.3 MASS, VOLUME, AND BULK DENSITY MEASUREMENTS

The total mass, volume, and bulk density (B.D.) of the material in the compost pile and the manure stockpile at the start and end of the trial are presented in Table 4.2.

Table 4.2 Initial and final dry mass, wet bulk density and volume of material

	Start (July 13, 2004)	Finish (October 21, 2004)	Change
Dry mass, (kg)			
Manure stockpile	19,754	17,750	(10.2%)
Compost pile	19,252	14,050	(27.2%)
Wet bulk Density, (kg m^{-3})			
Manure stockpile	531	480	(9.6%)
Compost pile	525	540	2.9%
Volume, (m^3)			
Manure stockpile	95.9	77.1	(19.6%)
Compost pile	100.1	50	(50.1%)

Parentheses indicate reduction

The final dry mass and volume of the compost (turned) and the manure stockpile (static) were less than the initial values (Table 4.2). At the end of the trial, the percentage reduction in dry mass from the compost pile was higher (27.2%) than percentage reduction from the manure stockpile (10.2%). The greater decrease in the mass of the compost pile was caused by greater loss of water and volatile solids. The volume of the compost pile was reduced by almost 50.1% whereas there was only a 19.6% reduction in the volume of the manure stockpile. The compost pile was turned frequently during the composting period. Turning increased the porosity, free air space and permeability of the pile by breaking manure lumps into uniform-sized granular particles. The small-sized manure particles helped not only to improve the aeration inside the pile but also the release of trapped heat and water vapor into the atmosphere (Haug 1995). At the end of the trial, the material in the manure stockpile was visually observed to contain many large lumps as compared with more uniform and smaller particles in the compost pile. Due to the presence of large lumps in the manure stockpile, the final B.D. was decreased by 9.6% from the initial B.D. However, there was a 2.9% increase in final B.D. of the compost pile from the initial value due to the presence of more uniform and smaller-sized particles.

The volumes of the compost and manure piles were also determined with a windrow profiler which measured the cross-sectional areas of the piles (Table 4.3).

Table 4.3 Volume estimated from the windrow profiler data

	Start (July 20, 2004)	**After reshaping (August 18,2004)**	**Finish (October 28, 2004)**	**Change***
	m^3	m^3	m^3	%
Static	84.9	N/A	67.6	(20.4)
Turned	92.6	64	45.8	(50.1)

***Parentheses indicate reduction**

The volumes of the compost pile (turned) and the manure stockpile (static) were reduced by 50.1% and 20.4%, respectively over the course of composting period (Table 4.3). The initial cross-section of the piles was measured one week after the formation of the piles. The volume estimation using a windrow profiler at the end of first week indicated that volume was reduced 11.5% and 7.5%, respectively from the compost and manure piles after first week of pile formation (Tables 4.2 and 4.3). On August 18, 2004, the compost pile was also scanned after reshaping. There was 30.8% reduction in the volume of the compost pile after one month of the trial. The reduction in the volume is one of the parameters used to assess the efficiency of the composting process (Rynk 1992). The high rate of decomposition results in higher volume and mass reduction as compared to static storage. The greater decrease in volume of the compost pile indicates that the rate of microbial activity was higher than in the manure stockpile. The turning of the compost pile played an important role in volume reduction. The greater porosity and smaller particles caused greater moisture losses from the compost pile. The manure stockpile was undisturbed, resulting in less favorable conditions inside the pile for active decomposition.

The cross-sections of the compost and manure piles at the beginning and end of the trial and the cross-section of the compost pile before and after reshaping are shown in Figures 4.2, 4.3, and 4.4. The cross-section is drawn using a third-degree polynomial trend line.

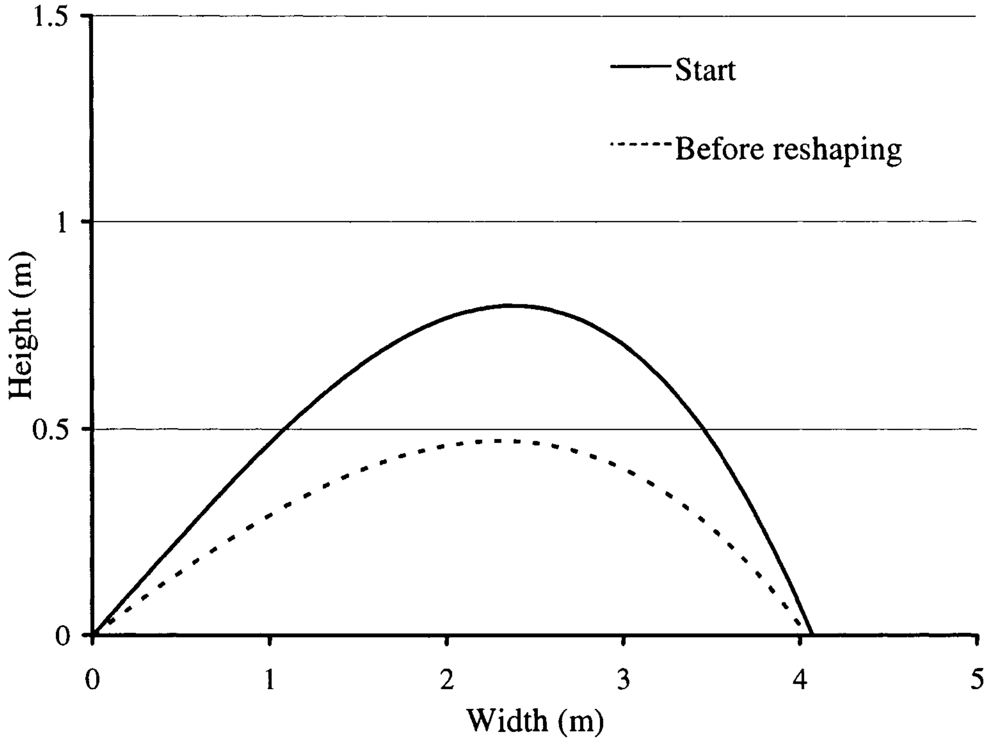


Figure 4.2 Cross-section of the compost pile at the start and before reshaping.

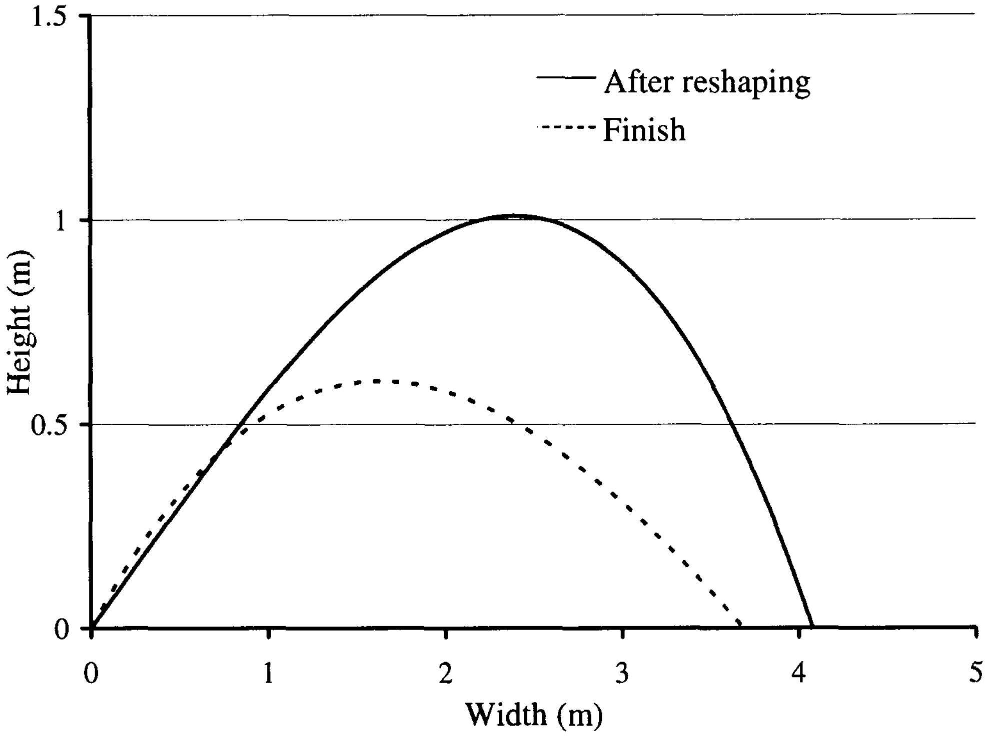


Figure 4.3 Cross-section of the compost pile after reshaping and at the finish.

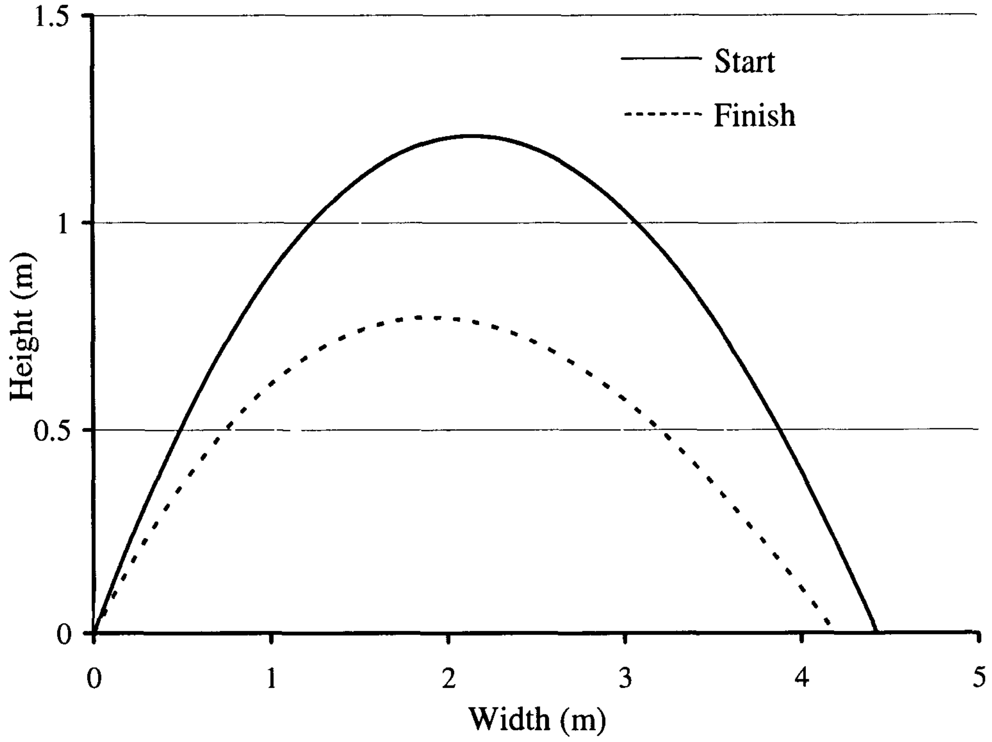


Figure 4.4 Cross-section of the manure stockpile at the start and finish.

4.4 TEMPERATURE DATA

The temperature of the piles was measured on each sampling day. The temperature histories of the compost pile and the manure stockpile at two different depths are shown in Figures 4.5 and 4.6.

The highest temperatures were measured during the first and second week of composting (Figures 4.5 and 4.6). As the composting process started the temperature in the manure stockpile decreased more rapidly than in the compost pile. The maximum temperature (68°C) was observed in the compost pile during the second week of composting. The temperature of the compost pile remained above 50°C until Day 36. During this phase, the high temperature was due to the generation of heat during the rapid decomposition of the organic matter.

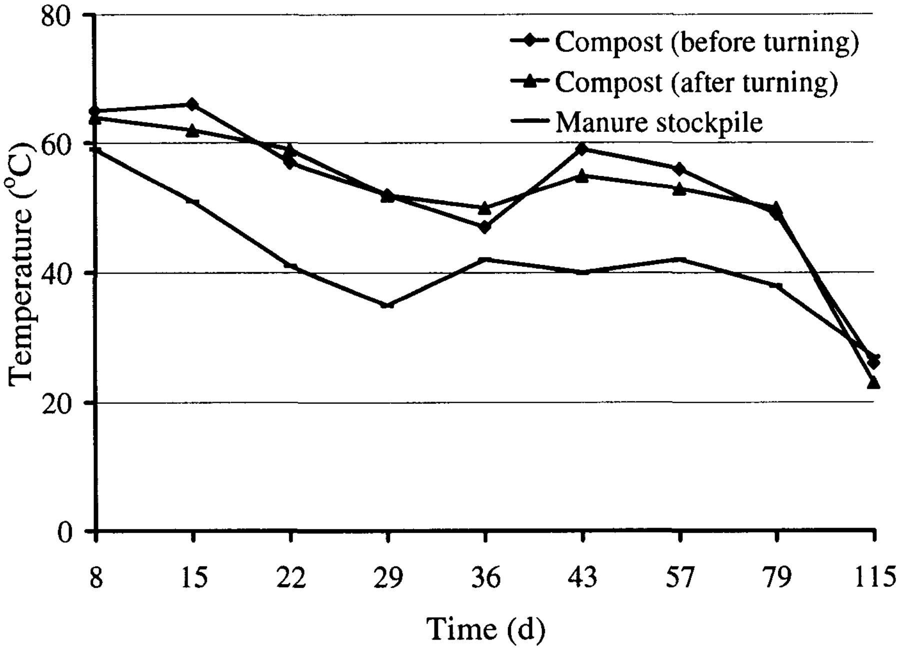


Figure 4.5 Temperature profile at 1 m depth.

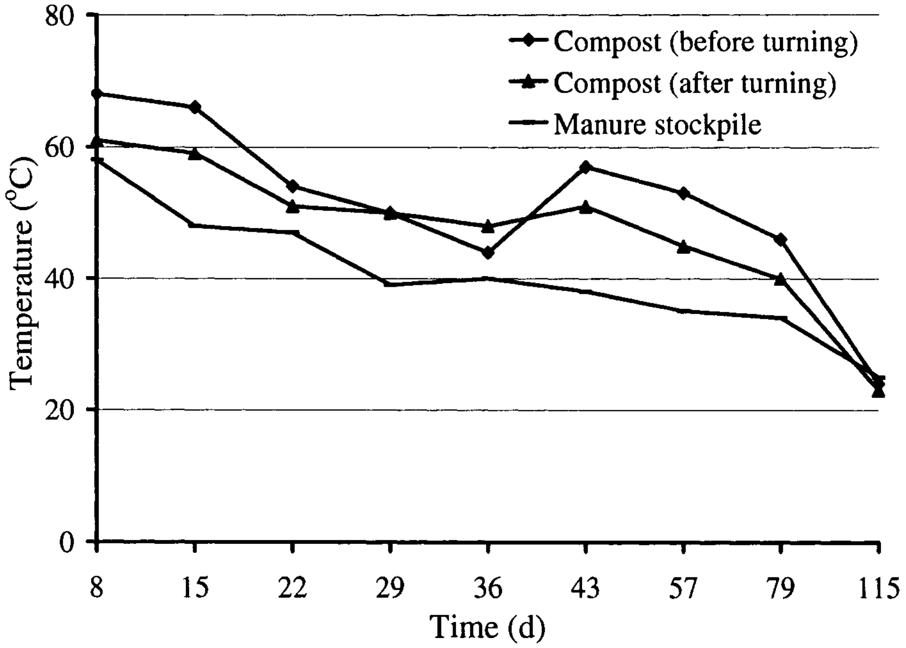


Figure 4.6 Temperature profile at 0.33 m depth.

After 30 days of composting, the compost pile had cooled down and the temperature decreased to 44°C (Figures 4.5 and 4.6). The weekly turning of the compost pile resulted in higher moisture losses from the compost pile than from the manure stockpile. The rapid volume reduction and higher temperature in the compost pile as compared from the manure stockpile indicated that the microorganisms were actively involved in the decomposition process during first month of the composting process (Table 4.3 and Figure 4.6). However, due to frequent turning of the compost pile, the recommended M.C. was not maintained in the pile. On Day 36, the compost pile was reshaped. The length of the pile was shortened and the material was piled to increase height and width. The temperature increased to 57°C on Day 43. The increased temperature might have resulted from better insulation resulting from lower area-to-volume ratio and increased microbial activity due to better aeration and increased availability of substrate and moisture. The temperature of the compost pile gradually declined to 24°C on the last day of the trial.

The temperature in the manure stockpile was always lower than in the compost pile (Figures 4.5 and 4.6). The lower temperature and the lower overall volume reduction in the manure stockpile indicated that the microbial decomposition of organic material was not as rapid as in the compost pile. Furthermore, the lower volume reduction in the manure stockpile at the end of the trial also indicated that the rate of decomposition was slower. The temperature remained at thermophilic levels (58°C) during the first 10 days of the trial but decreased more rapidly than in the compost pile.

4.5 CARBON TO NITROGEN RATIO

Table 4.4 shows the nutrient characterization of the compost pile and the manure stockpile as analyzed by Enviro-Test Laboratories, (Saskatoon, SK) and AAFC (Lethbridge, AB).

Table 4.4 Nutrient characterization (based on initial dry mass basis).

	Total Carbon	Elemental C		Total Nitrogen	Elemental N		Carbon:Nitrogen ratio
	%	Total kg	kg Mg^{-1} dry mass	%	Total kg	kg Mg^{-1} dry mass	
Compost (ENV)*							
Initial	22.2	4274	222	1.71	329	16.9	13:1
Final	16	2248	116	1.49	209	11.1	11:1
Change	(27.9%)	(2026)	(106)	(12.8%)	(120)	(5.8)	(15.3%)
Compost (AAFC)*							
Initial	28	5391	280	1.84	354	18.5	15.2:1
Final	17.1	2403	127	1.6	225	11.8	10.7:1
Change	(38.9%)	(2988)	(153)	(13.1%)	(129)	(6.6)	(29.6%)
Manure (ENV)							
Initial	20.8	4109	212	2.13	421	21.2	9.8:1
Final	25.2	4473	233	2.03	360	18.6	12.4:1
Change	21.2%	364	21	(4.7%)	(61)	(2.6)	26.5%
Manure (AAFC)							
Initial	25.5	5037	260	1.86	367	18.2	13.7:1
Final	23.6	4189	220	2.36	419	21.7	10:1
Change	(7.5%)	(848)	(40)	26.9%	52	3.5	(26.3%)

Parentheses indicate reduction/Losses

*** ENV = Enviro-Test Laboratories, AAFC = Agriculture and Agri-Food Canada**

Material samples were initially sent to Enviro-Test Laboratories for chemical analysis, but some ambiguity was observed in the results. The increase in C:N ratio in the manure stockpile experiment over the course of the trial were observed, however, no previous studies reported such a trend. To verify these results, samples that had been stored at 4°C in sealed containers were retested at AAFC, Lethbridge (Table 4.5). The results of the chemical analysis were different and the latter analysis seemed to be more consistent with previous studies conducted at the same location with similar beef feedlot manure (Hao *et al.* 2002). Therefore, data were interpreted using the results of the AAFC analysis.

The initial C:N ratios of the compost pile and the manure stockpile were 15.2:1 and 13.7:1 respectively (Table 4.4). The initial C:N ratios were considered to be low, as an initial C:N ratio of 25-40 has been suggested as optimum for composting (Diaz *et al*, 1993; Rynk 1992). At a low C:N ratio, the readily-available carbon can be quickly utilized by microorganisms without stabilizing the nitrogen and excessive nitrogen is lost to the atmosphere in the form of ammonia (NH_3). The elemental carbon present initially in the compost and manure was also found to be low 28.5% and 25.4%, respectively of total initial dry mass. However, several studies indicate that initial carbon content of manure varies between 40-50% of total dry mass (Fontanive *et al.* 2004; Rynk 1992). The possible reason for a low C:N ratio and carbon content was mixing of soil with the manure during scraping operation. Manure was scraped using a front end loader from the non concrete floor of a beef feedlot.

The final C:N ratios of the compost pile and the manure stockpile were reduced by 29.6% and 26.3%, respectively, from the initial values (Table 4.4). Several previous studies have also reported a loss of carbon and nitrogen due to active decomposition of organic matter by microorganisms (Michel *et al.* 2004; Fontanive *et al.* (2004).

It was also observed that reductions in TC, TN, and the C:N ratio of the manure stockpile were lower than those of the compost pile (Table 4.4). In the manure stockpile, there was only 848 kg loss of total carbon on a dry matter basis at the end of the trial as compared to 2988 kg of total carbon loss in the compost pile. The low reduction of nutrients from the manure stockpile indicates the slow rate of microbial decomposition of organic matter.

The nitrogen losses were 129 kg on a dry matter basis at the end of the trial in the compost pile as compared to a 52 kg gain of nitrogen in the manure stockpile. However while performing the analysis, there is (±10%) inherent error associated with the measurements of TN content and estimations of dry mass using moisture contents (AAFC). While taking the associated errors into the consideration, change in nitrogen at the end of trial was considered zero. Several studies have reported the TN loss during composting as high as 50% (Witter and Lopez-Real 1988; Sanchez-Monedero *et al.* 2001). Most of the N is lost either as NH_3 or N_2 into the atmosphere through volatilization due to dry conditions and high evaporation rate. The high temperature and high wind speed and the negligible rainfall at the experiment site throughout the trial, the possibility of leachate formation and lost of N as a nitrates under sub surface is very minimal.

4.6 END PRODUCT APPEARANCE OF THE PILES

During spreading, the end product from the compost pile was observed to be more uniform and to contain aggregates of about 2 mm diameter, whereas the manure from the stockpile was composed mostly of large lumps (150 mm diameter). The end product from the compost pile was brownish and had an earthy smell. The manure from the stockpile, however, had a green color and a strong odor of ammonia, indicative of incomplete oxidation of N compounds. The thick crust on the surface of the manure stockpile might have trapped odorous gases inside the pile and prevented aeration. In the compost pile, however, regular turning resulted in a more uniform structure, a comparatively drier end-product and less entrapment of gases than in the manure stockpile.

4.7 BIOGENIC GREENHOUSE GAS EMISSIONS

Figures 4.7 - 4.12 present the estimated hourly fluxes on the sampling days and the cumulative GHG (CO_2, CH_4, and N_2O) emissions from the compost pile and the manure stockpile.

4.7.1 Carbon dioxide emissions

The mean hourly CO_2 fluxes from the compost pile and the manure stockpile ranged from 3 -148 g m^{-2} h^{-1} and 6-113 g m^{-2} h^{-1}, respectively (Figure 4.7). The lowest CO_2 emission rates were observed on the day that the piles were established. However, the highest CO_2 emissions were observed on Day 43 from the compost pile (148 g m^{-2} h^{-1}) and on Day 2 in the manure stockpile (113 g m^{-2} h^{-1}) (Figure 4.7).

Off-gases were collected immediately after the turning of the compost pile. Both piles emitted CO_2 throughout the composting period. However, CO_2 emissions from the compost pile remained higher than the manure stockpile throughout the trial. The favorable M.C. inside the pile during the first three weeks of the trial from the compost pile resulted in same CO_2 emissions. However, after one month of the trial, a thick crust on the surface of the manure stockpile might have lowered gases emissions into the atmosphere. On Day 22, dramatically lower CO_2 emissions were observed compared with the previous week's emissions, even though the ambient air conditions (temperature and wind speed) were found to be quite similar on Days 22 and Day 15 (Table 4.1). A possible explanation could be that the old sample vial caps were used on Day 22 due to a delay in the shipment of new caps. The manufacturer recommends not to perforate the caps with hypodermic needles more than twice. There is a possibility that gases might have escaped the vial before analysis, since the old caps had already been perforated several times.

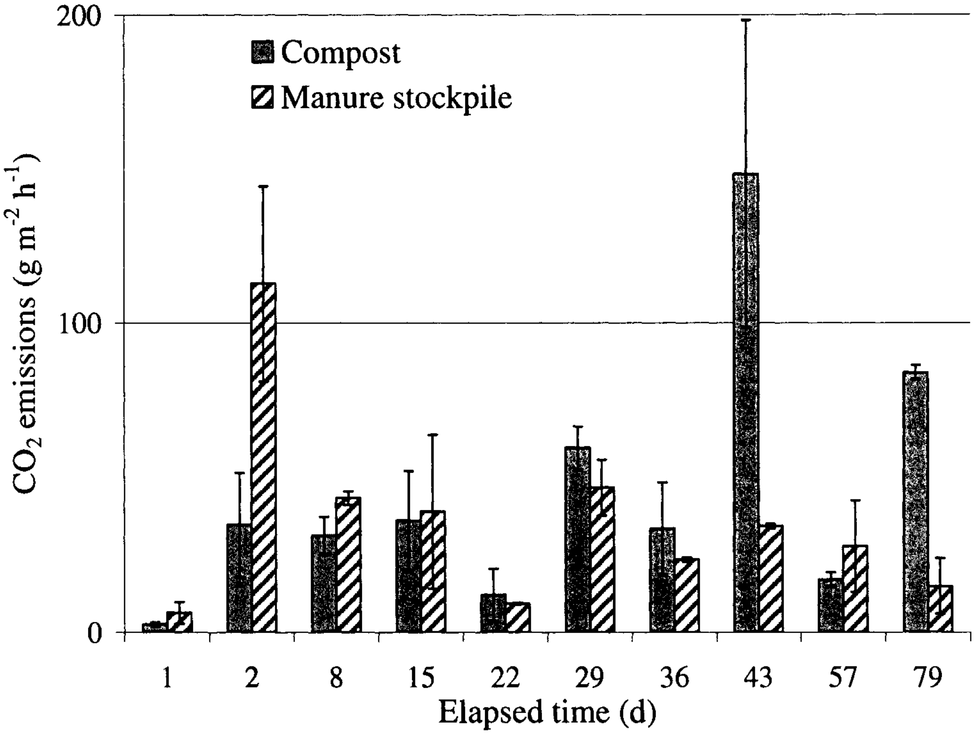


Figure 4.7 Carbon dioxide emissions

(Standard error bars indicate variation among three zones)

The immediate rapid increase in CO_2 emissions after Day 1 indicated that microbial decomposition was high during the initial stages in both the compost pile and the manure stockpile (Figure 4.7). During the mid stage of the composting process (after Day 36), comparatively higher CO_2 emissions were observed from the compost pile. This could have been due to the reshaping of the compost pile and addition of water. On Day 36, the length of the compost pile was reduced and the material was piled to increase the width and height of the pile. The reshaping of pile might have increased the microbial activity by redistributing and exposing substrate. Possibly as a result of this increased microbial decomposition, a sharp temperature peak was observed on Day 43 of the trial (Figures 4.5 and 4.6). The manure stockpile, on the other hand, was undisturbed and CO_2 emissions from the manure stockpile remained relatively low after Day 29. This clearly indicates the strong effect of turning on CO_2 emissions from the compost pile. Turning incorporates fresh air into the compost pile and increases the gas diffusion rate by increasing the porosity of material. However, due to the static nature of the manure stockpile, a thick layer of dry manure formed on the surface. This caused the development of unfavorable environmental conditions for aerobic microorganisms inside the unturned manure stockpile and hence, slower decomposition and comparatively low CO_2 emissions.

Figure 4.8 presents the comparison of cumulative carbon as CO_2 from the compost and manure piles. The total CO_2 emissions from the compost pile are higher than those from the manure stockpile. This is in an agreement with the results of other studies (Hao *et al.* 2001). The treatments (turned vs. unturned) had a significant ($P< 0.05$) effect on CO_2 emissions. Table 4.5 presents SAS results of significance with different factors.

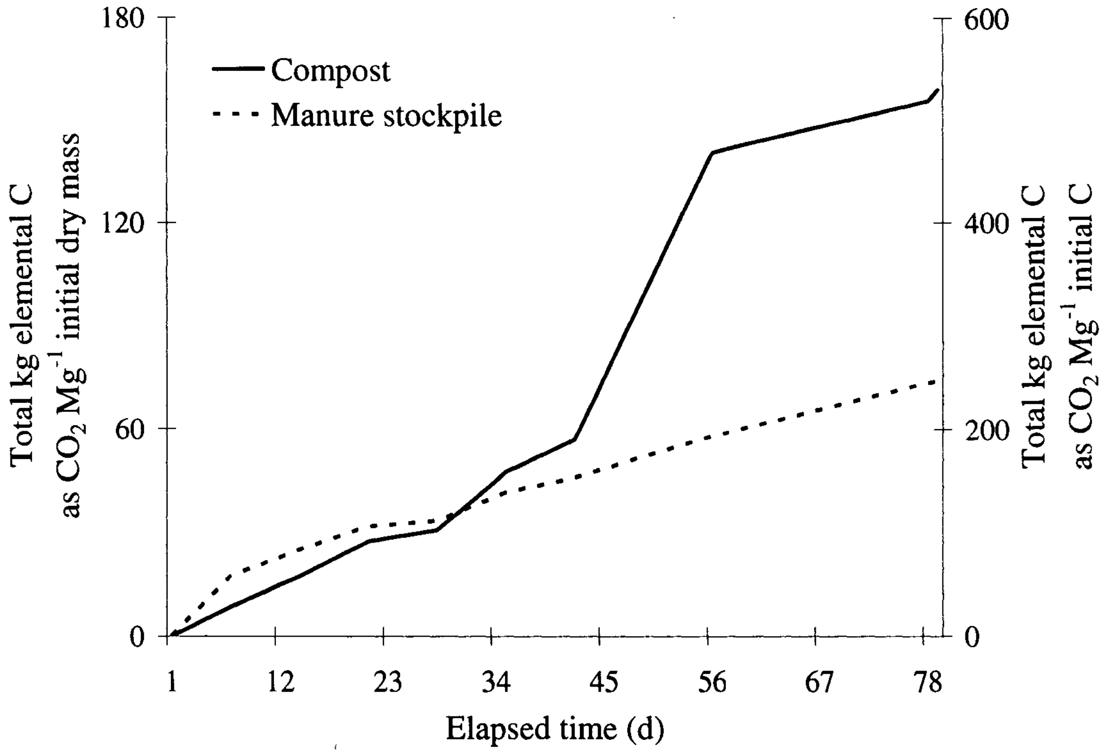


Figure 4.8 Cumulative carbon as carbon dioxide emissions (Assuming uniform emissions over total area and total elapsed time)

Table 4.5 Test of significance of carbon dioxide emissions

Effects	Num DF	Den DF	F value	Pr > F
Trt	1	20	896	<0.0001
Day	6	20	14321	<0.0001
Block	1	20	12081	<0.0001
Trt*Day	6	20	8493	<0.0001
Trt*Block	1	20	10869	<0.0001
Day*Block	2	20	3334	<0.0001
Trt*Day*Block	2	20	4659	<0.0001

4.7.2 Methane emissions

The CH_4 emissions were lower from the compost than from the manure pile. Methane emissions from the compost were detected only during the initial and final stages of composting (Figure 4.9). No CH_4 emissions were observed from the compost after Day 8 until Day 36 of the trial.

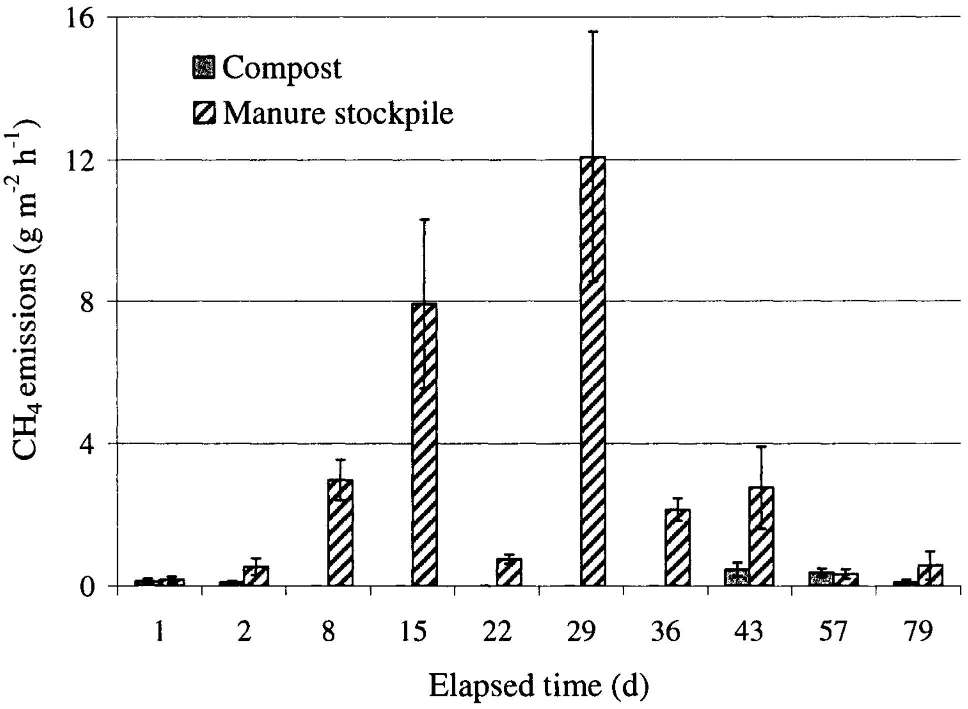


Figure 4.9 Methane emissions

(Standard error bars indicate variation among three zones)

The detection of CH_4 emissions from the compost during the initial stage of composting was quite similar to the patterns observed by Fukumoto *et al.* (2003) and Sommer and Møller (2000). The lack of emissions from Day 8 until Day 43 indicates the effect that turning had on the emission pattern. The compost pile was turned on a weekly basis during the first month after pile formation and once every two weeks after the first month of composting. The turning of the material favored development of aerobic conditions inside the compost pile. By turning the compost pile, the surface material was incorporated into the pile while material at the bottom was exposed at the surface. Therefore, the chances of CH_4 production due to the development of anaerobic zones inside the compost pile were minimal. Nevertheless, small CH_4 emissions were observed Day 43 onwards. A possible explanation could be the occurrence of anaerobic microsites inside the pile, resulting from compaction and reshaping of the compost pile. The compost pile was reshaped on Day 36 by reducing the length of pile and increasing the height and width. Regular turning of the compost resulted in a more uniform and smaller-sized aggregates. The compactness and bulk density increased in the compost pile during the trial (Table 4.2) and thus some anaerobic microsites might have developed inside the pile. On Day 36, 6000 L of water were added to the compost pile and measurable CH_4 emissions were observed the following week. The water addition might have created wet conditions inside the pile which are favorable for CH_4 generation. It is also suspected that the bigger pile size and interrupted air flow through the compost pile after reshaping might have caused small methane emissions. The results are consistent with the findings of Fukumoto *et al.* (2003), where a large compost pile (0.9 m height and 2 m diameter) showed high CH_4 emissions as compared to smaller compost pile (0.7 m height and 1.4 m diameter) due to the existence of some anaerobic sites in the large pile. Fukumoto *et al.* (2003) in their study emphasized that the piling scale of manure is an important factor in gas emission rates. As the size of manure piles increased, the number and size of anaerobic sites also increased. He *et al.* (2000) also stated that, even under aerobic conditions, anaerobic micro-sites may still exist inside aggregates.

Comparatively high CH_4 emissions were detected from the manure throughout the trial (Figure 4.10), indicating the presence of an anaerobic environment inside the manure stockpile. Methane emissions peaked at 12.5 g m^{-2} h^{-1} on Day 29. The lack of turning restricted air flow and moisture losses, creating an anaerobic environment conducive to CH_4 production. Hellebrand (1998) also concluded that inefficient aeration favors the development of anaerobic sites inside the pile. The CH_4 emission pattern from the manure stockpile agrees well with the findings of previous studies (Sommer and Møller, 2000; Hellmann *et al.* 1997). Lower CH_4 emissions on Day 22 might be ascribed to using the faulty sample vial caps and the windy conditions. The high ambient temperature (23.9°C) and wind velocity (14.2 km h^{-1}) on Day 22 compared to the weather conditions on previous sampling day (Table 4.1) could have affected the emission rates. The effect of wind speed on gas sampling was also discussed by Teshima (2003).

Figure 4.10 shows the comparison of cumulative carbon as CH_4 emissions from the compost and manure. Table 4.6 presents SAS results of significance with different factors. The total CH_4 emissions from the manure stockpile were significantly ($P < 0.05$)

higher than those from the compost. The interaction between time and CH_4 emissions from the compost and manure also was significant ($P < 0.05$) effect.

Table 4.6 Test of significance of methane emissions

Effects	Num DF	Den DF	F value	Pr > F
Trt	1	20	9866	<0.0001
Day	6	20	1809	<0.0001
Block	1	20	192	<0.0001
Trt*Day	6	20	1846	<0.0001
Trt*Block	1	20	40	<0.0001
Day*Block	2	20	107	<0.0001
Trt*Day*Block	2	20	271	<0.0001

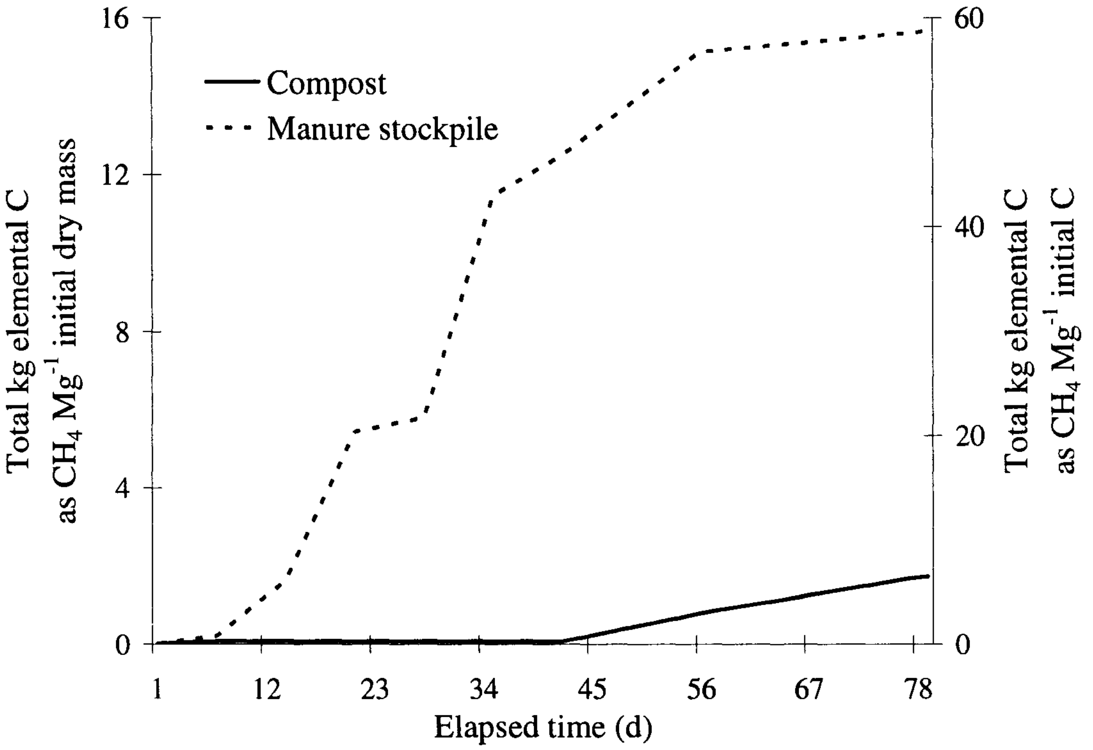


Figure 4.10 Cumulative carbon as methane emissions

(Assuming uniform emissions over total area and total elapsed time)

4.7.3 Nitrous oxide emissions

Compared with average CO_2 and CH_4 emissions from the piles, N_2O emissions from compost and manure were relatively low. However, N_2O emissions were less variable throughout the trial period (Figure 4.11). The small standard errors also indicate relatively small variation between N_2O emissions from the three different zones.

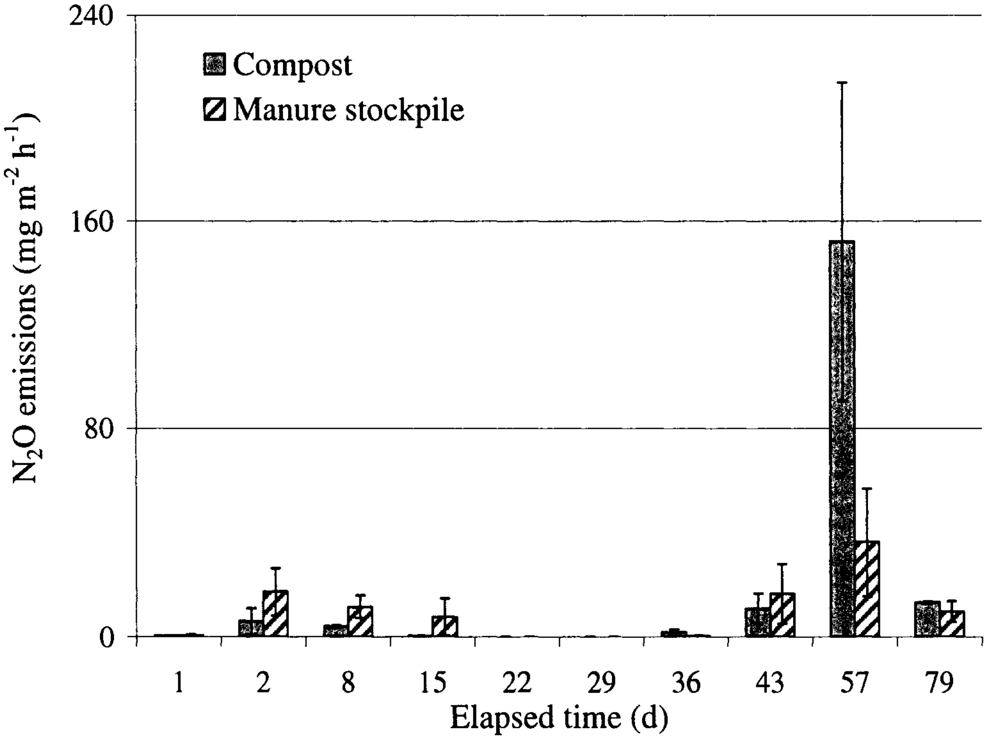


Figure 4.11 Nitrous oxide emissions

(Standard error bars indicate variation among three zones)

The maximum N_2O emissions were 152 mg m^{-2} h^{-1} and 36 mg m^{-2} h^{-1} on Day 57 from the compost pile and the manure stockpile, respectively. Only Day 21 and Day 29 did not show any N_2O emissions. Some sampling error is suspected on these days. On Day 22 and Day 29, old sample vial caps were used for the gas sampling due to delay in the shipment of new caps. The manufacturer recommends not to perforate the caps more than twice. There is a possibility that gases might have escaped during storage.

In this study, the pattern of N_2O emissions rates observed during the trial were in agreement with the results of the study conducted by Hao *et al.* (2001) and Hellmann *et al.* (1997). Several previous studies found that the N_2O emissions from livestock waste are influenced by the initial C:N ratio, moisture content, temperature, aeration, type of feedstock, compost age, pile depth, and the shape of the pile (Hellebrand 1998 and Hao *et al.* 2001). The initial C:N ratio in this experiment was very low (Table 4.3), the recommended range for effective composting being between 25 and 40:1 (Rynk 1992). Due to the low C:N ratio, the lack of organic carbon could have played an important role in N_2O leakage from nitrification and denitrification reactions during the initial phase of decomposition, since N_2O is an intermediate product during of nitrification and denitrification (Ballestro and Douglas 1996). When organic carbon is not available in sufficient amounts in the material, the transformation of organic nitrogen into ammonia (NH_3) takes place. During the oxidation of NH_3 to nitrate, N_2O gas can be produced as a by-product during nitrification by autotrophic bacteria. He *et al.* (2001), in their study on GHG emissions from aerated composting, found an abrupt increase in N_2O emissions immediately after the depletion of available organic carbon.

During the later stages of the trial (from Day 36 onwards), an increasing trend in N_2O emissions from both the compost and the manure was observed (Figure 4.11). This is similar to the pattern observed by He *et al.* (2001), in which peak N_2O emissions occurred at the end of the composting process. Denitrification might be a cause of N_2O gas production during the latter stages of composting when more anaerobic sites develop. Nitrous oxide gas is emitted as a byproduct during the denitrification of nitrate (NO_3^-) into nitrogen (N_2). The denitrification process is mainly influenced by oxygen concentration, temperature, and moisture content. The compost pile was reshaped on Day 36 and subsequently more water was added the compost pile. It is, therefore likely that the addition of water created anaerobic sites inside the pile. The turning of the compost pile also increased the compaction and bulk density of the material. The occurrence of anaerobic micro-sites and consequently N_2O emissions through denitrification were likely greater in the deeper and more compacted compost pile during latter stages of the trial (after Day 36). However, the formation of a thick surface layer of manure on the unturned manure stockpile might have restricted the air flow into the pile.

Figure 4.12 compares cumulative nitrogen as N_2O emissions from the compost and the manure. Table 4.7 presents SAS results of significance with different factors. Significant difference ($P< 0.05$) was found among N_2O emissions from the compost and the manure.

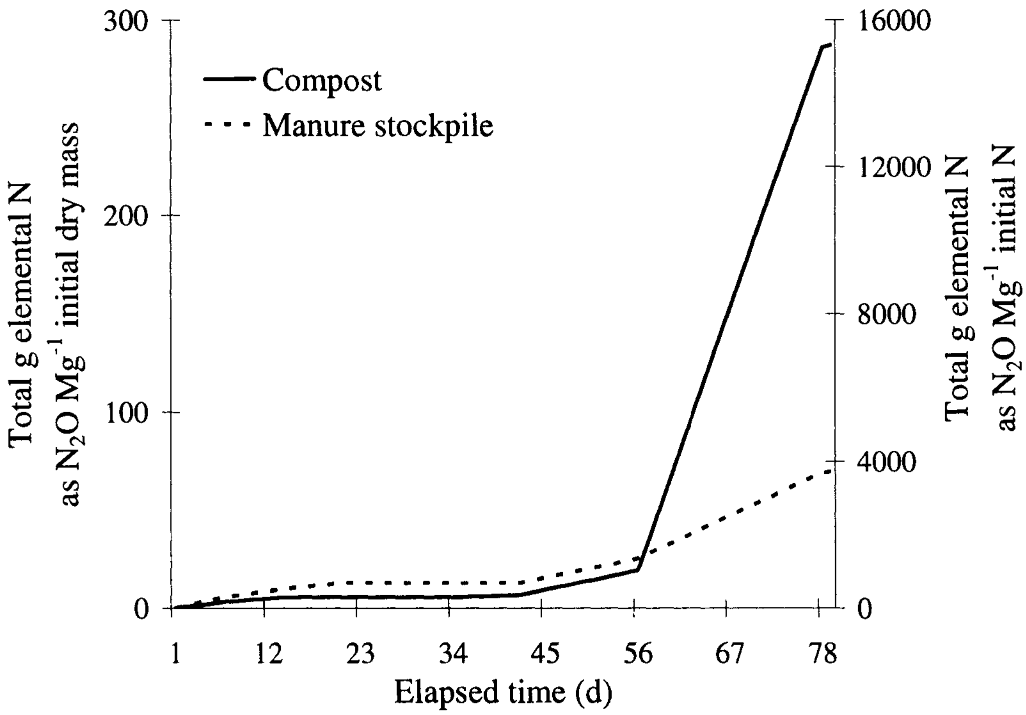


Figure 4.12 Cumulative nitrogen as nitrous oxide emissions (Assuming uniform emissions over total area and total elapsed time)

Table 4.7 Test of significance of nitrous oxide emissions

Effects	Num DF	Den DF	F value	Pr > F
Trt	1	21.9	211	<0.0001
Day	6	21.9	104	<0.0001
Block	1	21.9	358	<0.0001
Trt*Day	6	21.9	94	<0.0001
Trt*Block	1	21.9	310	<0.0001
Day*Block	2	21.9	281	<0.0001
Trt*Day*Block	2	21.9	210	<0.0001

4.8 MASS BALANCE FOR GREENHOUSE GAS EMISSIONS AND CARBON AND NITROGEN

Table 4.8 presents TC and TN losses as CO_2, CH_4, and N_2O in compost and manure piles using different methodologies.

Table 4.8 TC and TN losses as carbon dioxide and methane and nitrous oxide

	Linear regression	Second order polynomial	Logarithmic model 5 min	Logarithmic model 10 min
Compost pile	kg Mg^{-1} initial dry mass			
C loss as CO_2	62	120	159	156
CH_4	0.7	1.2	1.7	1.4
Total C loss	62.7	121.2	160.7	157.4
N loss as N_2O	0.07	0.14	0.28	0.11
Manure stockpile				
C loss as CO_2	28	53	64	76
CH_4	7.4	13.5	15.3	15.6
Total C loss	35.4	66.5	79.3	91.6
N loss as N_2O	0.032	0.04	0.07	0.06

Greenhouse gas emissions are estimated using the total surface area of the piles. In this study, GHG emissions were assumed to be uniform from the whole pile. However, several researchers found the high spatial variability in emissions indicating that the maximum emissions occur at the top of the pile on GHG emissions (Fernandes *et al.* 1994; Wolter *et al.* 2004). Therefore, during further discussion of mass balances of carbon and nitrogen, the assumption of uniform emissions from whole surface area is redefined. Secondly, GHG emission rates from half an hour interval were assumed to be true representative of daily GHG emission rates. Greenhouse gases were collected for half an hour interval at the same time on selected sampling days throughout the trial. The emission rates estimated from half an hour time interval were extended over the 24 h interval. It is very unlikely that emission rate would remain constant due to diurnal change in weather conditions (temperature, rainfall and humidity) which could affect the microbial activity.

4.8.1 Carbon balance as Carbon dioxide and methane

Total carbon losses as CO_2 and CH_4 using different methodologies are documented in Table 4.8. The mass balance comparison has been made between calculated TC losses using AAFC results (Table 4.4) and measured TC losses as gaseous content using logarithmic model suggested by Hutchison and Mosier (1981) at 5 min interval (Table 4.8). The calculated TC losses using AAFC results were 153 and 40 kg Mg^{-1} initial dry mass at the end of trial from the compost and manure pile (Table 4.4). The total gaseous C losses as CO_2 and CH_4 are measured 160.7 and 79.3 kg Mg^{-1} initial dry mass from compost and manure pile using Hutchison and Mosier (1981) (Table 4.8). A comparison could be made between change in measured carbon content and TC losses as CO_2 and CH_4 by redefining the assumption of total emitting surface with one half of the total surface area as an emitting surface. By assuming 50% of the total surface area as an emitting surface, TC losses as CO_2 and CH_4 are measured 80.5 and 39.6 kg Mg^{-1} initial dry mass as compared to TC losses of 153 and 40 kg Mg^{-1} initial dry mass at the end of trial from the compost and manure pile. The difference in measured C and calculated C might be due to sampling error on Day 22 and 29. The relatively lower emissions on these days resulted in lower cumulative CO_2 emissions.

It is also suspected that small fraction of C can possibly be lost in the form of volatile low chain fatty acids such as acetate, and was not measured in this study. Several studies suggested that during the formation of CH_4 under anaerobic conditions, some C can be lost in the form of volatile acids such as acetic acid and butyric acid (Speece 1996). These volatiles acids are the intermediate component of fermentation during methanogenic process.

4.8.2 Nitrogen balance as nitrous oxide

Total nitrogen losses as N_2O from the compost were 2.1% of calculated TN losses at the end of trial by considering 50% of the total surface area as an emitting surface (Table 4.4 and 4.8). The measured N losses as N_2O using Hutchison and Mosier (1981) from the compost pile are 0.28 kg Mg^{-1} initial dry mass (Table 4.8) as compared to calculated 6.6 kg-N Mg^{-1} initial dry mass loss at the end of trial (Table 4.4). Several studies have reported the similar pattern of as low as 0.5% N loss as N_2O emissions during composting (Hao *et al.* 2004; Sommer 2001; Kuroda *et al.* 1996). In this study, N lost as N_2O are calculated 1% of TN with the assumption of one half of total emitting surface of the compost pile. Most of the N is lost through volatilization either as NH_3 or N_2. Due to the negligible rainfall and warm and windy atmospheric conditions during the trial, the possibility of leachate formation and loss of N as nitrates under sub surface is very minimal. Most of the N might be lost in the form of NH_3 due to high wind speed (14 km h^{-1}) throughout the trial. In manure stockpile, the measured N loss as N_2O was 0.035 kg Mg^{-1} initial dry mass (Table 4.8). Table 4.4 shows net gain of N at the end of trial. However, while performing the analysis, there is (±10%) inherent error associated with the measurements of TN content and estimations of dry mass using moisture contents (AAFC). While taking the associated errors into the consideration, change in nitrogen at the end of trial could be considered not different from zero. The small amount of N losses as N_2O emissions during the study indicated that most of the N is lost through volatilization either as NH_3 or N_2 which is in agreement with the finding of previous studies (Hao *et al.* 2004; Sommer 2001).

4.9 NON-BIOGENIC GREENHOUSE GAS EMISSIONS

Non-biogenic GHG emissions were estimated indirectly by measuring the amount of diesel fuel consumed during the establishment of the piles, turning of the compost pile, and hauling and spreading the end products on the field.

4.9.1 Fuel consumption during material acquisition and pile formation

The total amounts of diesel fuel consumed during material acquisition and the establishment of the piles of compost (19 t) and manure stockpile (18.8 t) were 31.9 L and 28.5 L respectively.

4.9.2 Fuel consumption during turning of the compost

The compost pile was turned once every week during the first month of the trial and once every two weeks thereafter. The manure stockpile was not turned. The amount of diesel fuel consumed by the tractor-operated turner was measured after each turning. Figure 4.13 presents the amount of fuel consumed in each turning operation.

The fuel consumption during the first turning was high, but less fuel was consumed during turning thereafter until Day 29. A possible reason for the high fuel consumption during the first turning was the high moisture content and the presence of large lumps of manure. The time taken to complete each turning was also noted and the first turning took more time than in subsequent weeks. The decrease in fuel consumption after the first turning is likely due to decreased moisture content and reduced volume of compost. On Day 36, after reshaping of the pile and the addition of water, the fuel consumption increased (Figure 4.13). It was also observed that, due to the increased width of the compost pile, the rotating drum of the turner took more time to penetrate the compost.

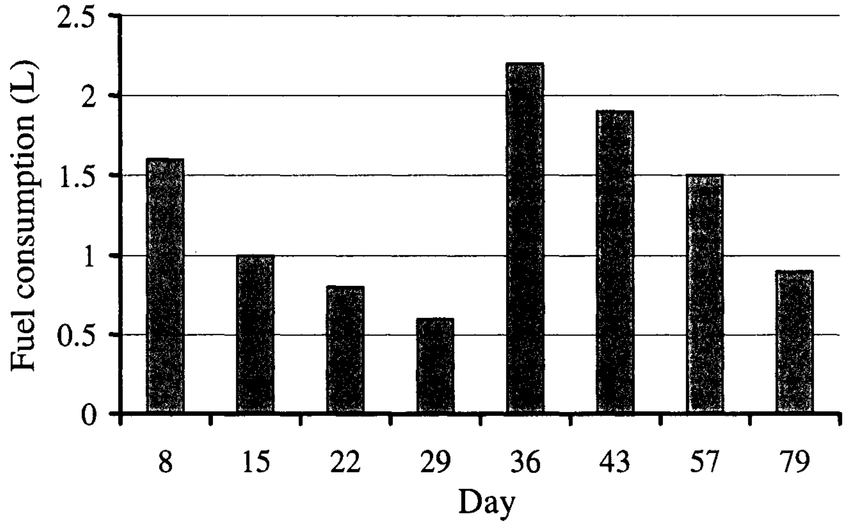


Figure 4.13 Fuel consumption in each turning operation

4.9.3 Fuel consumption during hauling and spreading

As expected, the amount of diesel fuel consumed (2.6 L Mg^{-1} initial dry mass) during the spreading of the manure was higher than the amount consumed while spreading the compost (2.1 L Mg^{-1} initial dry mass) due to the high moisture content of the manure, low bulk density, and smaller reduction in mass and volume. At the end of the trial, there was only a 27.3% reduction in the mass of manure compared with a 48.6% mass reduction in compost (Table 4.2). The end product of the compost pile was dry and uniformly-structured compared to the large wet lumps present in the manure stockpile. The large wet lumps were observed to impede the rotating tynes of the spreader and consequently, more fuel was consumed.

4.10 TOTAL GREENHOUSE GAS EMISSIONS FROM NON-BIOGENIC SOURCES

Table 4.9 presents the amount of fuel consumed and the related non-biogenic GHG emissions. The total GHG emissions are estimated per ton (Mg) of initial dry mass.

Table 4.9 Non-biogenic greenhouse gas emissions (Per Mg initial dry mass).

	Fuel consumed	CO_2	CH_4	N_2O	Equivalent CO_2
	L Mg^{-1}		g Mg^{-1}		kg Mg^{-1}
Compost Pile					
Hauling and pile formation	1.69	4614	0.44	0.68	4.83
Turning	0.55	1502	0.15	0.22	1.58
Hauling and spreading	2.1	5733	0.55	0.84	6.04
Total	**4.34**				**12.5**
Manure Stockpile					
Hauling and pile formation	1.53	4177	0.40	0.61	4.4
Hauling and spreading	2.6	7098	0.68	1.1	7.4
Total	**4.13**				**11.8**

4.11 NET GREENHOUSE GAS EMISSIONS COMPARISON

Figure 4.14 presents net biogenic GHG emissions from the compost pile and manure stockpile by assuming 50% of the total surface area as an emitting surface. The equivalent GHG emissions were analyzed using Proc Mean statement (SAS 2001). The standard error bars indicates the high variability in GHG emissions among three zones. Biogenic GHG emissions from the compost were higher (385 kg-CO_2 equivalent Mg^{-1}

dry mass) than from the manure (369 kg-CO_2 equivalent Mg^{-1} dry mass) mainly due to CO_2 release in the compost pile (Figure 4.14). Methane contributed almost 64% of total kg-CO_2 equivalent emissions from the manure as compared with only 7% of total kg-CO_2 equivalent emissions from the compost. The high CH_4 emissions from the manure stockpile were mainly due to the anaerobic conditions developed in the pile due to lack of turning. Nitrous oxide contributed 17% and 4.5% of total kg-CO_2 equivalent emissions from the compost and the manure stockpile. The statistical analysis was performed on three biogenic GHG datasets. The treatments, compost and manure pile, were significantly different ($P<0.05$) with respect to carbon dioxide, methane and nitrous oxide.

Figure 4.15 presents net non-biogenic GHG emissions from the compost pile and manure stockpile. The overall non-biogenic GHG emissions associated with compost were slightly higher than those from the manure (Figure 4.15). No P values and standard error bars are associated with the analysis of non-biogenic emissions due to the lack of replications. The statistical analysis was not performed on the data set from non-biogenic analysis. As compared to the biogenic GHG emissions, however the non-biogenic GHG emissions were small. It is worth mentioning that regular turning of the material contributed about 13% of non-biogenic emissions associated with the compost, whereas spreading and hauling of residuals contributed 63% of non-biogenic emissions from the manure stockpile compared to 48% from the compost windrow.

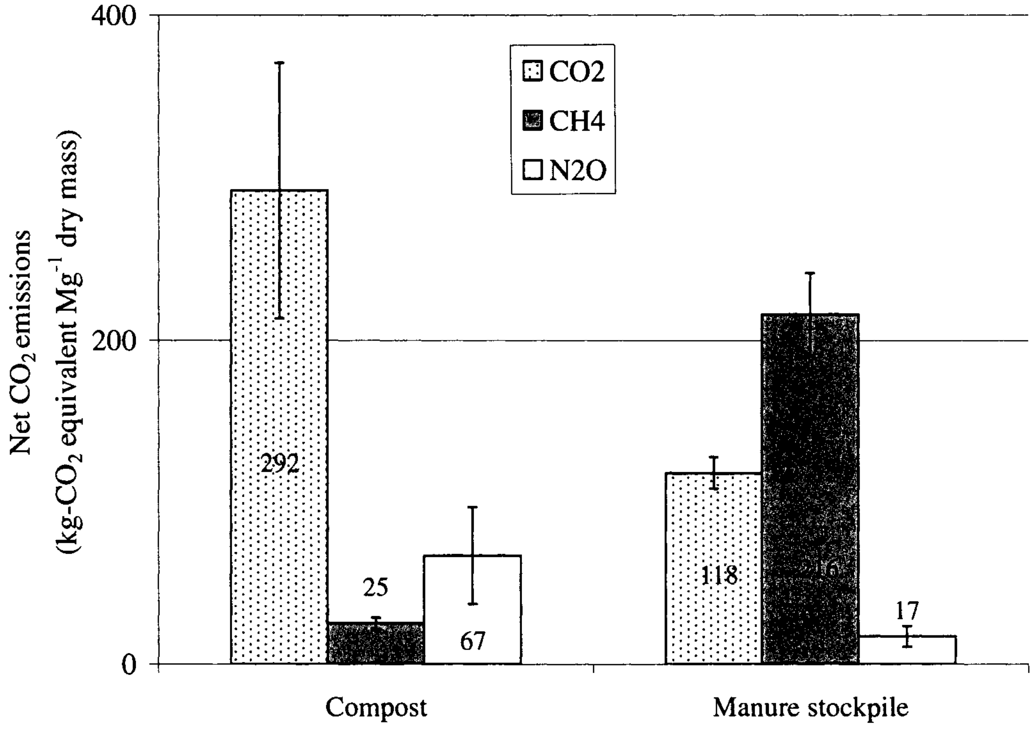


Figure 4.14 Biogenic greenhouse gas emissions

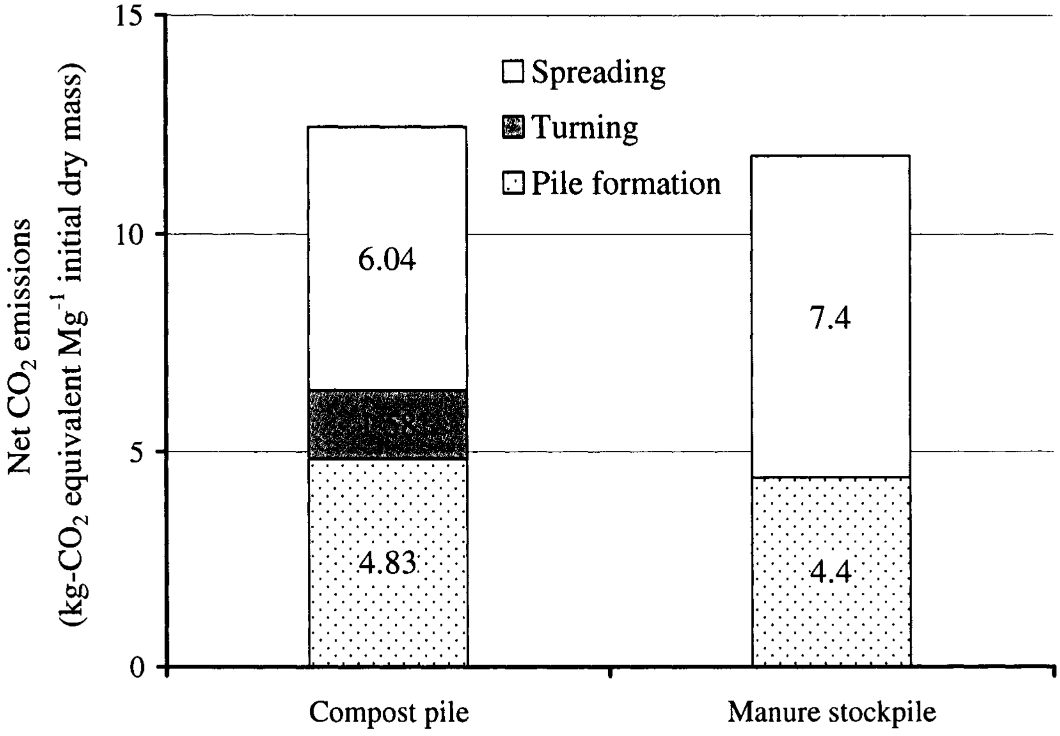


Figure 4.15 Non-biogenic greenhouse gas emissions

It is worth mentioning that IPCC guidelines suggested that biogenic CO_2 emissions arising from natural decomposition should not be counted in GHG inventories as it is a part of carbon cycle (IPCC 2001 (b)). Biogenic carbon is part of the natural carbon balance and it will not add to atmospheric concentrations of carbon dioxide. In accordance with the IPCC guidelines, exclusion of biogenic CO_2 emissions from the compost and manure stockpile result in significantly higher ($P<0.05$) GHG emissions (CH_4 and N_2O) from manure pile (233 kg-CO_2 equivalent Mg^{-1} dry mass) as compared to compost pile (92 kg-CO2 equivalent Mg^{-1} dry mass).

Chapter 5. Conclusions and Recommendations

5.1 CONCLUSIONS

The total GHG emissions from the compost pile and the manure stockpile were estimated to be 397 and 364 kg CO_2-equivalent Mg^{-1} initial dry mass, respectively based on uniform GHG emissions over time and whole pile using Hutchison and Mosier (1981) model at 5 min interval. From this study it can be concluded that the total GHG emissions from the compost pile were 8.3% higher than those from the manure stockpile. The higher biogenic GHG emissions from the compost pile (384 kg CO_2-equivalent Mg^{-1} initial dry mass) as compared to 352 kg CO_2-equivalent Mg^{-1} initial dry mass from the manure stockpile were mainly due to biogenic CO_2 release. The CH_4 emissions in terms of CO_2 equivalent contributed 64% of biogenic GHG emissions from the manure stockpile and only 7% from the compost windrow. The lower CH_4 emissions from the compost pile were mainly due to the aerobic conditions maintained in the pile through frequent turning during the course of the trial. The pattern of GHG emissions from two treatments are in agreement with the available literature (Hao *et al.* 2001). The GHG emission rates are comparable with several literature sources (Czepiel 1996; Hellmann 1997).

The total non-biogenic GHG emissions from the compost pile and the manure stockpile were 12.5 and 11.8 kg CO_2-equivalent Mg^{-1} initial dry mass, respectively. Higher non-biogenic emissions occurred from the compost, primarily associated with fuel combustion during the turning of the compost pile. Turning contributed almost 13% of total non-biogenic GHG emissions from the compost. Turning emissions from the compost pile were balanced with lesser emissions during the transportation and spreading of compost residual. Non-biogenic GHG emissions from both the piles, however, were small compared with the biogenic GHG emissions.

Overall, the compost showed higher peak temperatures (67°C) compared to the manure stockpile (59°C), smaller and more uniform aggregates from the compost windrow at the end of trial, and greater mass reduction (48.6%) as compared to the manure stockpile (27.4%).

5.2 RECOMMENDATIONS AND SUGGESTIONS FOR FUTURE WORK

This study was an attempt to analyze the biogenic and non-biogenic GHG emissions from two different manure management processes. To date, uncertainty and variability are associated with the data collection methodology, and in the quantification and interpretation of the GHG data. The standardization of GHG collection and analytical methodologies should be addressed.

It would be impractical for farmers to voluntarily adopt GHG mitigation techniques in livestock waste management unless they improve profitability. Currently, GHG emission reductions are not of economic consequence to producers. The direct and indirect benefits associated with applying composted residuals, however, might contribute to improved profitability as a result of reduced volume, less fuel combustion in spreading, and better aggregate structure.

Greenhouse gas emissions from livestock waste management systems might influence climate change. Thus, potential GHG mitigation options for livestock waste management, such as aerobic composting, could be considered as an alternative to conventional manure storage. The initial high investment to procure machinery involved in maintaining aerobic conditions must be taken into account, however, and, as mentioned, there are currently no financial incentives available to encourage livestock waste management practices that reduce GHG emissions.

It is also worth mentioning that IPCC guidelines suggested that biogenic CO_2 emissions arising from natural decomposition should not be counted in GHG inventories as it is a part of carbon cycle (IPCC 2001 (b)). Biogenic carbon is part of the natural carbon balance and it will not add to atmospheric concentrations of carbon dioxide. In accordance with the IPCC guidelines, emission factors for biogenic CO_2 emissions are considered zero. The exclusion of biogenic emissions would result in significantly higher ($P<0.05$) GHG emissions from manure pile as compared to compost pile.

More intensive research to quantify GHG emissions from manure management with replications under different environmental conditions would be helpful in understanding the variation of the GHG emissions.

It is also important to consider the whole management system when evaluating GHG mitigation options for handing livestock waste, including emissions from soil resulting from undecomposed C and N compounds after the application of residuals. The application of residuals from the manure stockpile might produce higher GHGs due to the presence of undecomposed C and N compounds resulting from the incomplete decomposition of material under anaerobic conditions inside the manure stockpile. To the knowledge of the author, no comprehensive study has been conducted so far which quantifies GHG emissions after the application of residuals.

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Appendix 1. Emission factors for ground vehicles

Vehicle type	Emission control technology	CO_2 (g/L)	CH_4 (g/L)	N_2O (g/L)
Gasoline				
Autos	New 3-way catalyst	2360	0.24	0.45
	Aged 3-way catalyst	2360	0.37	1.65
	Oxidation catalyst	2360	0.54	0.45
	Non-catalyst	2360	1.03	0.23
	Uncontrolled	2360	1.03	0.23
Light duty trucks	New 3-way catalyst	2360	0.37	0.45
	Aged 3-way catalyst	2360	0.48	1.65
	Oxidation catalyst	2360	0.46	0.45
	Non-catalyst	2360	0.87	0.23
	Uncontrolled	2360	0.87	0.23
Heavy duty trucks	3-way catalyst	2360	0.28	1.65
	Non-catalyst	2360	0.48	0.23
	Uncontrolled	2360	0.76	0.23
Diesel				
Light duty vehicles	Advanced controlled	2730	0.067	0.4
	Moderate controlled	2730	0.045	0.4
	Uncontrolled	2730	0.075	0.4
Heavy duty vehicles	Advanced controlled	2730	0.2	0.4
	Moderate controlled	2730	0.2	0.4
	Uncontrolled	2730	0.21	0.4
Off-road vehicles (Tractors, etc.)				
Gasoline		2360	0.19	0.23
Diesel		2730	0.26	0.4

Environment Canada 2002

Appendix 2. SAS Code for test of significance

```
options formdlim='*';
DATA Carbon;
INPUT Trt$ Zones day block obs carbon;
CARDS;


comp       1      1      1      1  2.354785714
comp       1      2      1      1  20.37188257
comp       1      8      1      1  22.01639325
comp       1     15      1      1  9.487397516
comp       1     22      1      1  1.563458269
comp       1     29      1      1  45.78394507
comp       2      1      1      2   3.32314868
..
;
PROC mixed;
CLASS trt Zones day block;
MODEL Carbon = trt day block trt|day|block/DDFM=kr outp = predicted;
Repeated day/subject= Zones (trt) type= ANTE(1) R RCORR;
repeated /group=zones;
LSMEANS trt day trt*day*block/pdiff;
run;
proc print data=predicted;
var resid;
run;
```

Appendix 3. SAS results

```
          The SAS System

            The Mixed Procedure

             Model Information

Data Set                 WORK.CARBON
Dependent Variable         carbon
Covariance Structure       Variance Components
Group Effect             Zones
Estimation Method          REML
Residual Variance Method    None
Fixed Effects SE Method     Prasad-Rao-Jeske-
                             Kackar-Harville
Degrees of Freedom Method   Kenward-Roger


           Class Level Information

 Class   Levels   Values

 Trt        2   Manure comp
 Zones      3    1 2 3
 day        7    1 2 8 15 22 29 44
 block      2    1 2


              Dimensions

    Covariance Parameters          3
    Columns in X                 60
    Columns in Z                  0
    Subjects                   120
    Max Obs Per Subject            1
    Observations Used            120
    Observations Not Used          0
    Total Observations           120
```

```
              Iteration History

 Iteration   Evaluations   -2 Res Log Like      Criterion

     0            1        966.97630548
     1            2        922.37479314      0.00007598
     2            1        912.65839382      0.00023246
     3            1        902.72577723      0.00070542
     4            1        892.45492118      0.00214710

               The SAS System                  The Mixed Procedure

              Iteration History

 Iteration   Evaluations   -2 Res Log Like      Criterion

     5            1        881.87546675      0.00657301
     6            1        871.06016171      0.02021458
     7            1        860.06673585      0.06217160
     8            1        848.88420097      0.18761499
     9            1        839.33399203      0.40367451
    10            1        834.05484793      0.00502887
    11            1        832.28269080      0.00067604
    12            1        832.04317482      0.00001710
    13            1        832.03752737      0.00000001
    14            1        832.03752262      0.00000000


            Convergence criteria met.


          Covariance Parameter Estimates

          Cov Parm     Group      Estimate

          Residual    Zones 1      0.3004
          Residual    Zones 2      851.91
          Residual    Zones 3     1357.16


               Fit Statistics

         -2 Res Log Likelihood         832.0
```

```
   AIC (smaller is better)        838.0
   AICC (smaller is better)        838.3
   BIC (smaller is better)        846.4


     Null Model Likelihood Ratio Test

       DF    Chi-Square      Pr > ChiSq

        2       134.94        <.0001


              The SAS System

             The Mixed Procedure

       Type 3 Tests of Fixed Effects

                 Num     Den
Effect            DF      DF    F Value    Pr > F

Trt                1      20     896.66    <.0001
day                6      20    14321.9    <.0001
block              1      20    12081.0    <.0001
Trt*day            6      20    8493.71    <.0001
Trt*block          1      20    10869.9    <.0001
day*block          2      20    3334.71    <.0001
Trt*day*block      2      20    4659.93    <.0001
```