

**WINTER COMPOSTING OF SEPARATED PIG SLURRY SOLIDS AND
GREENHOUSE GAS EMISSIONS**

by

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ABSTRACT

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One strategy to manage pig slurry is centrifugation and composting of the solids fraction to produce a value added product to distribute manure nutrients further from production sites. This study determined turned windrow composting was suitable for processing slurry solids throughout winter. It was also the first attempt at combining automated chambers and a Fourier Transform Infrared spectroscopy analyzer to measure multiple gases during the composting process; the system proved capable but captured fluxes better if conducted in an area sheltered from wind. Straw and woodshavings were shown suitable as bulking materials for composting slurry solids, however, the lack of porosity provided by woodshavings created anaerobic conditions that doubled the greenhouse gas emissions compared to those of straw, 1,126 kg CO₂-equivalent Mg⁻¹ compared to 526 kg CO₂-equivalent Mg⁻¹. Either bulking material produced compost of quality for use in agricultural or soil blending applications and was free of manure pathogens.

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1.0 INTRODUCTION

In Manitoba, most of the agricultural soils are phosphorus (P) deficient and require P addition; however, in the south-eastern municipalities of La Broquerie and Hanover, a high density of pig production facilities has resulted in excessively high soil P on land located within a close proximity to the barns because it was costly to transport slurry (Statistics Canada, 2006). Build up of P in soil can be an issue because it may run-off into waterways during spring melt events and causes eutrophication in lakes (Kumaragamage et al., 2011). In an effort to decrease the amount of P loading into lakes, Manitoba's regulations for manure application rates changed from residual soil nitrogen (N) based limits to residual soil P based, these changes came into effect in November 2013. The regulation also prohibits spreading manure during the winter, from November 10 to April 10, making manure management and storage over the winter a challenge. These regulations leave some pig producers with too much liquid pig slurry and not enough land base or storage to stay in compliance and they will need to evaluate new management options suitable for their operations. A manure treatment option, that large pig producers may be interested in, involves using a centrifuge to separate the slurry into a liquid fraction and a "solids" fraction. The liquid fraction has a higher N content that can continue to be applied on land close to the pig production site, whereas the solid fraction, also referred to as separated pig slurry solids, contains most of the P and can be more efficiently transported and distributed further away because the material weighs less and has lower volume. However, the resulting slurry solids have a low carbon to nitrogen ratio; therefore, if the slurry solids were stored or directly spread onto a field then they would be prone to N losses through ammonia volatilization or nitrate leaching and would be quite odorous (Dinuccio et al., 2008; Hansen et al., 2006; Sorensen and Rubaek, 2012). Further composting of the slurry solids could reduce odours as well as convert the available ammonium into stabilized

organic N. This would allow the compost product to be stored, without losing nutrients through volatilization or leaching, until the compost can be applied to land (Larney et al., 2006).

Composting such a unique product such as slurry solids has not been a common practice in Manitoba, therefore there was a need to evaluate this approach before it was recommended and implemented by producers. Generally, on-farm operations prefer turned windrow composting because costs are relatively low and it requires less labour from the producer to manage the process compared to other aerated, rotating drum, or in-vessel composting technologies (Environment Canada, 2013; Rynk, 1992). However, turning the compost in the winter with a windrow turner can expose the material to low air temperatures making it difficult to maintain thermophilic conditions required for the composting process and to meet the Canadian Council of Ministers of the Environment (CCME) guidelines for pathogen elimination, a requirement for product quality (CCME, 2005). Additionally, slurry solids are high in N and moisture, and have a uniform texture; feedstock materials with these types of characteristics can be prone to excessive greenhouse gas and nitrogen gaseous losses if the composting process was not properly maintained. For example, methane, CH₄, a greenhouse gas can be produced if anaerobic conditions are present (Ahn et al., 2011; Sommer and Moller, 2000; Hao et al., 2001; Hao et al., 2004). Nitrous oxide (N₂O) can also be produced through incomplete nitrification or denitrification when anaerobic conditions exist leading to additional greenhouse gases as well as nitrogen losses (Ahn et al., 2011; Fukumoto et al., 2003; Fukumoto and Inubushi, 2009; Fukumoto, 2012; Hao et al., 2001; Hao et al., 2004). Other sources of gaseous nitrogen losses could include ammonia (NH₃), nitric oxide (NO), and nitrogen dioxide (NO₂). Losses of nitrogen gases can be dependent on compost management practices such as turning frequency and timing, moisture content and available carbon in the bulking material (Ahn et al., 2011; Elwell et al., 2002; Fukumoto et al., 2003; Fukumoto and

Inubushi, 2009; Georgaki et al., 2009; Hao and Chang, 2001; Hao et al., 2004; Hellman et al., 1997; Jiang et al., 2011; Maeda et al., 2009; Paillat et al., 2005). Emissions of nitrogen gases not only result in loss of nitrogen that could be applied to soil in the final compost product, but NH_3 and nitrogen oxide (NO_x) gases in the atmosphere can also contribute to acid rain, air smog and deplete stratospheric ozone (Krupa, 2003; Hao and Chang, 2001).

Previous studies that have measured gaseous losses from turned windrow composting have used static vented chambers and focused only on monitoring CO_2 , CH_4 , and N_2O during the composting process to determine best management practices (Hao et al., 2004; Hellman et al., 1997). However, there were limitations with the use of static vented chambers to measure additional gases, and the laborious and time consuming data collection limits the frequency of sampling, providing poor temporal coverage of gas emissions (Sommer et al., 2004; Anderson et al., 2010). Technology for measuring gas emissions from turned windrows has advanced, allowing for the capability to also measure NH_3 , NO_2 and NO emissions. An innovative technology that combines an automated gas flux chamber system (Long-term Automated Chambers, LICOR BioSciences Inc., Lincoln, NB) in series with a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer (DX4015, Gasmeter Technologies Inc., Helsinki, FIN) had not been used to measure gas emissions during the composting process before, but it had the capability to offer automated monitoring of multiple gases including CO_2 , CH_4 , NO_2 , NH_3 , NO , and NO_2 on a nearly continuous basis. Monitoring all these gases simultaneous could give a better understanding of the effect of management practices and the interaction of these gases during the process of windrow composting of slurry solids. Having the ability to measure gaseous emissions can investigate not only if composting separated solids is feasible but also allows determining management practices, for example which bulking material is better suited for limiting greenhouse gas and nitrogen

gaseous emissions. In addition, recommendations on the type of bulking material can be given based on ability to provide insulation throughout the winter, duration of composting process, and overall quality of the finished product.

This being the first attempt to use automated chambers and FTIR system for measuring emissions from compost windrows standard operating procedures and potential sources of error needed to be determined. The automated chamber and FTIR system has advantages over static vented chambers but it still relies on gas diffusion into the chamber headspace to calculate gas fluxes. This has been identified as a problem when wind moves horizontally through the windrows, reducing the ability to capture gas fluxes. However, with the increased frequency of gas flux measurement, a relationship between wind speed and reduced gas flux can be established. Additionally, to give an indication of the ability of the automated chamber and FTIR system to account for total gaseous losses the Total C and N flux emissions calculated by the automated chambers and FTIR system can also be compared against Total C and N losses calculated by mass loss balance.

Therefore, the objectives of this thesis were to:

- 1) Determine standard operating procedures and potential sources of error for measuring greenhouse gases and nitrogen gases during the composting process using a newly developed automated chamber and FTIR system.
- 2) Determine the potential for composting separated pig slurry solids in winter and overall quality of the finished compost product.
- 3) Determine if the emission data from the automated chamber and FTIR system can be used to detect differences in management practices by being capable of associating composting properties and practices such as temperature, oxygen, moisture content and turning

frequency with greenhouse gas and nitrogen gaseous losses throughout the composting process.

- 4) Determine the effect of bulking agent when used to compost separated pig slurry solids including greenhouse gas and nitrogen gaseous losses, ability to provide insulation throughout the winter, duration of composting process, and overall quality of the finished product.
- 5) Compare the losses of Total C and N flux emissions calculated by the automated chambers and FTIR system against Total C and N losses calculated by mass loss balance to determine the ability of the automated chamber and FTIR system to account for total losses.
- 6) Observe the relationship of wind speed on reducing measurement of gas fluxes during windrow composting.

Chapter 2 describes the standard operating procedures that were developed for the automated chamber and FTIR system to be used for measuring gas fluxes from turned windrow composting. The settings that were selected for the FTIR analyzer were tested against standard gases in the laboratory to assess the accuracy of identifying and quantifying gases. A series of laboratory analyses were conducted on NH_3 to identify and minimize potential sources of sorption. The behaviour of the gases from actual gas fluxes that were measured from compost windrows were also analyzed to verify that an Excel Macro that was developed for selecting data points for use in linear gas flux calculations could be used.

Chapter 3 was the first field experiment conducted; the purpose was to determine the suitability of composting separated pig slurry solids throughout the winter to produce a value

added product. The automated chamber and FTIR system was used to measure gaseous fluxes; this information was used to determine if the automated chamber and FTIR was capable of detecting differences in management practices and the fluxes could be explained by properties of the compost windrow. The Total C and N flux emissions calculated by the automated chambers and FTIR system were also compared with those calculated by mass loss balance to determine the ability of the automated chamber and FTIR system to account for total losses.

Chapter 4 presents results from the second field experiment, which compared straw and woodshavings as bulking materials for composting separated pig slurry solids. Factors considered for determining which bulking material was more suitable included greenhouse gas and nitrogen gaseous losses, ability to provide insulation throughout the winter, required duration of composting process in order to produce mature compost, and overall quality of the finished product. The Total C and N flux emissions calculated by the automated chambers and FTIR system were also compared with those calculated by mass loss balance to determine the ability of the automated chamber and FTIR system to account for total losses when measured in a sheltered area.

Chapter 5 highlights the potential effect wind had on the underestimation of gas emissions when using the automated chamber and FTIR system by comparing carbon dioxide fluxes measured in Chapters 3 and 4 versus wind speed. Potential factors that increased the effect of wind on underestimation were also identified.

Chapter 6 provides an overall synthesis and summarizes recommendations for using the automated chambers and FTIR system, the suitability of composting separated pig slurry solids throughout the winter to produce a value-added product, and best management practices for minimizing greenhouse gas and nitrogen gaseous losses.

2.0 DEVELOPMENT OF AN AUTOMATED SYSTEM FOR DETERMINING GREENHOUSE AND NITROGEN GAS FLUXES FROM COMPOST WINDROWS

2.1 Abstract

Combining automated flux chambers with a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer can offer advantages over static vented chambers such as nearly continuous in-situ multi-gas analysis. However, this being the first attempt to measure gaseous emissions during the composting process with this technology, the overall setup needed to be described and verification of gas concentrations and interferences, ammonia sorption, and behaviour of gases needed to be analyzed to determine potential limitations or inaccuracies. It was determined that the FTIR was capable of accurately identifying and measuring multiple gases including carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) at concentrations that may be emitted from compost windrows. The FTIR overestimated values of ammonia (NH₃), nitrogen dioxide (NO₂) and nitric oxide (NO) compared to standard gas concentrations; possibly because the gases may be more reactive; but there was consistency in overestimating these gases. However, because gas flux was determined as a rate consistent overestimation in concentration should not be an issue. The FTIR may not accurately measure N₂O when CO₂ was present because there could be an interference that the FTIR cannot account for. This should be acknowledged but it may not be a relatively important source of error as the fluxes from compost windrows are less sensitive compared to if fluxes were measured from soils. Laboratory testing indicated the potential for NH₃ adsorption onto surfaces as the NH₃ was purging from the system as well as potential for sorption in water vapour as condensation on metal surfaces. The type of tubing material, Teflon or BevA, did not make a difference in reducing the effect of NH₃ sorption; therefore, the extension tubing

remained as BevA. The similar behaviour of CO₂, CH₄, and N₂O gases when measured by the automated chamber and FTIR system from actual compost windrows allowed for the use of an Excel Macro developed to automatically select data points for use in gas flux calculations. The selection of data points for NH₃ and NO had to be checked manually for accuracy due to inconsistent linear accumulation in concentration over time. The automated chamber and FTIR system had few limitations that could not be overcome by following recommendations for setup and data analysis. It was determined that there was potential for this system to be used for measuring greenhouse gas and nitrogen gaseous emissions to develop best management practices for composting. The frequency of data collection reduces the need for gap filling compared to static vented chambers, allowing confidence in data collection especially with rapidly changing composting conditions. The ability to measure NH₃ and NO gases in addition to N₂O, provides insight on the dynamics of N losses, something static vented chambers are not able to do.

2.2 Introduction

2.2.1 Background

During the composting process, greenhouse gases (GHG's), carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O), as well as nitrogen (N) gases, ammonia (NH₃), nitrogen dioxide (NO₂) and nitric oxide (NO) can be produced. The quantity and timing of the gases produced vary by stage of the composting process and physical and chemical conditions that exist (Ahn et al., 2011, Amlinger et al., 2008; Fukumoto et al., 2003; Hao and Chang, 2001; Hao et al., 2004; Larney et al., 2006; Paul et al., 2001; Zhang et al., 2013). Mismanagement of the composting process can lead to excessive GHG emissions and N gaseous losses, which can be harmful to the environment

by contributing to climate change, acid rain, smog, or ozone depletion as well as reduce the amount of nitrogen in the final compost product (Larney et al., 2006; Hao and Chang, 2001; Krupa, 2003). Recently, pig producers are in search of best management practices for composting pig slurry solids in order to consider it as an alternative management practice. With a fairly unique product such as separated pig slurry solids it is important to identify the types and quantities of gases produced to understand what factors can influence excessive losses, for example high moisture can result in anaerobic conditions that produce CH₄ (Environment Canada, 2013).

Previously, most research that measured greenhouse gases from turned windrow composting utilized static vented chambers. Advantages of static-vented chambers are that they are inexpensive, do not require significant expertise, and the number of chambers used can easily be adapted to fit the criteria of the experimental site and design (Denmead, 2008; Brummell and Siciliano, 2011). However, a major disadvantage of static vented chambers was the limited amount of data collection because sample collection was laborious and time consuming which resulted in poor coverage of temporal variation in fluxes during rapidly changing composting processes (Sommer et al., 2004; Anderson et al., 2010). Spatial positioning of the static vented chambers may also lead to sampling errors when gases are measured from compost windrows because often in compost windrows, there are preferential airflow pathways or horizontal airflow caused by wind that can lead to significant fluxes from one area and not another (Sommer et al., 2004; Anderson et al., 2010). Other disadvantages of static vented chambers included being prone to human error and bias during sampling, delays in receiving ex-situ data analysis results making it hard to justify abnormal results, and difficulty in extracting enough sample gas for multi-gas analysis without creating a pressure differential within the chamber (Hutchinson and Livingston, 2001; Denmead, 2008; Brummell and Siciliano, 2011).

2.2.2 Automated Chamber and FTIR System

Recently, technology has been developed that may alleviate some of the limitations of static vented chambers, but has not yet been tested on windrow composting. Combining automated flux chambers (Long-term Automated Chambers, LICOR BioSciences Inc., Lincoln, NB) in series with a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer (DX4015, Gaset Technologies Inc., Helsinki, FIN) created a re-circulating loop gas monitoring system that allowed for unattended, systematic, nearly continuous, multi-gas analysis.

The automated flux chamber and FTIR system may be an improvement on static-vented chambers for monitoring GHG and N losses from compost but has not yet been deployed. The system still applied the same general principle of static vented chambers, which was to establish a linear relationship of gas accumulation with time over a specific area to calculate gas fluxes. To use the system, a PVC pipe collar was inserted into the surface of the compost windrow and an automated flux chamber unit was placed over the collar. When the automated flux chamber sealed over a collar embedded in the compost windrow, a re-circulating loop between the automated flux chamber and FTIR analyzer was established. Gases from the compost windrow diffuse to the chamber headspace based on the concentration gradient and air-filled porosity of the compost according to Fick's First Law (Denmead, 2008; Davidson et al., 2002). The FTIR was used to measure the concentration of the gases in the chamber headspace as air re-circulated between the chamber and FTIR. The accumulation in gas concentration versus time for a re-circulating loop was a sigmoidal curve, similar to a static chamber (Denmead, 2008). Initially, there was an exponential increase in gas concentration as the sample air mixed with the air in the chamber headspace, tubing, and analyzer. This was then followed by a linear increase in concentration,

referred to as the “linear accumulation period” and was used in the flux calculation. If the chamber remained closed, the concentration would level off and become constant because high gas concentrations would inhibit the gradient between the air in the compost and air in the chamber headspace (Denmead, 2008). Inclusion of non-linear exponential accumulation at the beginning of the observation can lead to an overestimation of gas emissions, whereas including a section where concentration was inhibited would result in an underestimation of gas emissions. When measuring multiple gases, data processing would be simpler if all gases measured behaved similarly and consistently because the selection of the linear accumulation period would be the same for each gas, and the selection of data points to be used in flux estimations can be automated. If the gases do not behave similarly and/or consistently, manual selection of the linear accumulation period would be required.

The automated flux chambers were computer programmed to open and close automatically at timed intervals, reducing sampling errors from human bias. The system used in this study included a multiplexer that accommodated up to eight chambers setup to run in sequence allowing for nearly continuous data collection. The continuous high frequency data can be used to get a better understanding of the dynamics and environmental factors involved in gas emissions (Brummell and Siciliano, 2011). The FTIR is a non-destructive analyzer, meaning the gas sample was not altered as it was analyzed, so the FTIR can be incorporated into the re-circulating-loop system and provide real-time, in-situ measurements (Brummell and Siciliano, 2011). Monitoring gas fluxes in real-time can allow a researcher to adjust measurement techniques or identify sampling errors immediately (Brummell and Siciliano, 2011). The FTIR was also capable of simultaneously measuring multiple gases that could potentially be emitted during the composting process, including CO₂, N₂O, CH₄, NH₃, NO₂, and NO. The FTIR used an interferometer to

produce a signal that contains all the frequencies in the infrared spectrum (Griffith and Galle, 2000). When this signal was passed through the sample gas the absorption and transmittance spectrum pattern at each frequency was produced, these spectra undergo Fourier transformation to identify the concentration of gases of interest present in the sample (Griffith and Galle, 2000). With the capability to measure gases simultaneously with the same instrument it can be easier to look at gas interactions and compare gas fluxes (Brummell and Siciliano, 2011). Although the automated flux chambers and FTIR system was likely an improvement over static vented chambers there may still be limitations. Multi-gas analysis required programming the FTIR to appropriate wavelengths for each gas to ensure there was no interference with other gas concentration determinations. However, in a personal communication with Dr. Richard Farrell (2013), Associate Professor at the University of Saskatchewan, it was suggested to investigate the interference of N_2O and CO_2 at conditions typical for compost windrows, despite the ability of the FTIR to account for potential interferences. In Dr. Farrell's experience using the FTIR and automated chamber system to measure soil gas fluxes, there was a slight decrease in N_2O when CO_2 was five times ambient conditions (Farrell et al, 2012).

Another possible concern with an automated chamber and FTIR system was error in NH_3 measurements because NH_3 was prone to sorption onto surfaces (Ferm et al., 2000; Harper, 2005; Mukthar et al., 2003). The majority of surface area in the chamber system was the extension tubing that was used to carry the gas from the chamber to the central multiplexer. The system was fitted with low-density polyethylene (LDPE) Bev-A tubing; however, Teflon tubing has been shown to reduce the sorption of NH_3 because it has an inert surface, has a low coefficient of friction, and low permeability to gases and water vapour (Baker and Mead, 2000; Ferm et al., 2000; Griffith and Galle, 2000; Mukthar et al., 2003). To potentially reduce the sorption of NH_3 the extension

tubing can be changed to Teflon. However, Teflon tubing was not as flexible as BeVA tubing so the tubing could be prone to kinking if not handled carefully in the field, which can cause damage to the tubing and lead to leaks. There are also other surfaces that could be major sources of NH_3 sorption when water condensation forms on them. The high water solubility and polarity of NH_3 can allow sorption to water condensation on surfaces (Harper, 2005). Sampling from compost windrows presents a concern for water condensation formation because the compost emits moist high temperature air that can produce humid conditions within sampling equipment that will condensate on cooler surfaces such as the aluminum manifold in the multiplexer or the steel dome of the automated chamber. If this was the case, changing the extension tubing to Teflon tubing may not make a difference.

Therefore, the objectives of this study were to:

- 1) Assemble and describe the FTIR and automated flux chamber system setup used on compost windrows.
- 2) Identify and give recommendations for reducing potential inaccuracies in gas flux determinations from using the above mentioned system caused by:
 - a. error in gas concentration due to FTIR settings
 - b. error in gas concentration due to interferences from other gases
 - c. ammonia sorption and condensation to surfaces of the system
 - d. behaviour of the gases delineating from linear accumulation

2.3 Materials and Methodology

2.3.1 Automated Flux Chambers and FTIR Setup

The volume of the automated flux chamber and FTIR system was 4,912.3 cm³, which included the automated chamber headspace (83% of the total volume), a multiplexer that allowed up to eight chambers, a Li8100a infrared gas analyzer (IRGA) (LICOR BioSciences Inc., Lincoln, NB), a FTIR analyzer, a Büchner flask, and tubing to connect the system in a re-circulating loop. Although the system did not require the Li8100a IRGA, it was decided to use it as a quick check to compare CO₂ concentrations against those measured by the FTIR. The Büchner flask was added as a precaution to catch water that accumulated in the tubing prior to reaching the FTIR sample cell. Water reaching the FTIR needed to be avoided because it would damage the sample cell, rendering it inoperable and would require costly replacement. Tubing with incoming air from the multiplexer was fed through a rubber stopper inserted in the top of the Büchner flask; gravity would have dropped any water droplets to the bottom of the flask, allowing air to continue through the side arm into the Li8100a IRGA. In the field, droplets of water were seen in the tubing up to 1 foot away from air exiting the chamber headspace, thus raised concerns; however, water droplets were never seen in the Büchner flask. The FTIR was recalibrated using inert N₂ standard gas every visit to the site, approximately every 2-3 days.



Figure 2.1 Setup of the multiplexer, Li8100a IRGA, FTIR, and data recording laptops in insulated storage unit. Büchner flask installed after the photo was taken.

The automated chambers were positioned on the peak of windrows. A 15 cm height, 23 cm diameter PVC pipe collar was inserted 8 cm into the surface compost windrow. The bases of the automated chambers were propped up with 1 x 1 inch lumber to keep the chambers from sinking in the compost. When the system was deployed, it was checked frequently for issues that prevented the chambers from closing properly. For example, NH_3 could corrode the metal pins, that connected the signal cable of the automated chamber with the multiplexer that signalled the opening and closing of the chamber, if the connection cover for the metal pins was not tightened enough to create a proper seal. Snow, ice and other debris also occasionally prevented the chamber

dome from opening and closing properly and needed to be cleared. Power outages also terminated the schedule of opening and closing of the chambers and required re-starting. The operation of the chambers also needed to be shut down if air temperatures were below $-30\text{ }^{\circ}\text{C}$ in order to prevent harm to any components (Gasmeter Technologies, 2009).



Figure 2.2 Automated chamber, in open position, setup on the peak of a compost windrow.

Eight automated chambers were available for use. To estimate gas fluxes, each chamber underwent a “measurement period” in which the concentration of gases was determined for a period of time. Each measurement period included a pre-purge, observation, and post-purge sub periods. For measuring gas fluxes from compost an observation period, the time when the chamber dome was closed, did not need to exceed 3 minutes in order to observe accumulation of gases emitted from compost windrows. Additionally, the total measurement period should be at least 30

minutes for each chamber, to allow proper pre- and post-purging of the system, time when the chamber dome was open and air is flowing through the tubing. Purging of the system was required to establish a steady gas concentration baseline and flush residual gases from the tubing.

During the observation period the chamber took 13 seconds to completely close and approximately 30 seconds for the air to mix within the system, resulting in an exponential accumulation in gas concentration. After this, a “linear accumulation period” occurred; to calculate a gas flux at least four consecutive data points were needed. The FTIR analyzed a gas sample and provided a data point of all gas concentrations every 10.5 seconds, which required the linear accumulation period to be at least 30-36 seconds. The process of selecting the four data points from the linear accumulation period was automated by creating an Excel Macro (APPENDIX A: Excel Macro Code for Observation Finder) that used the recorded time stamp from the closing of the automated chamber. In the setup used, there was one laptop computer that programmed the schedule of the automated chambers and there was another laptop computer to record the FTIR gas concentration analysis. There was a slight time drift between the two computers that required frequent synchronization to align the two time stamps. It would have been beneficial to have had only one computer running both software systems.

To calculate the gas flux for a chamber the following equation was used (Hutchinson and Livingston, 1993):

$$F = \frac{dC}{dT} \times \frac{V}{A} \times \frac{M}{V_{mol}} \quad (1)$$

where F is the gas flux ($\text{g C or N m}^{-2} \text{ sec}^{-1}$), $\frac{dC}{dT}$ is the linear rate of change in gas concentration during the linear accumulation period ($\text{mol mol}^{-1} \text{ s}^{-1}$), V is the total system volume (m^3), A is the

surface area of the chamber collar (m^2), M is the molecular weight of the gas ($g\ mol^{-1}$) and V_{mol} is the volume of a mole of the gas ($m^3\ mol^{-1}$) at the air temperature during the time of sampling.

2.3.2 Gas Measurement Testing

2.3.2.1 FTIR Settings and Verification of Gas Concentration Calibration

The FTIR required spectral references for each gas, obtained from Gaset Technologies Inc., to identify and quantify gases in the sample gas. The spectral references reflect the pattern of absorption and transmittance of infrared light across the wavelength range of 900 to 4,200 cm^{-1} ; essentially it is a “finger print” for each gas. Multiple reference spectra were required to cover the range of gas concentrations expected in gas samples. The FTIR stored the spectral reference concentrations in the software “library” and referred to these to compare against the gas sample spectra. Some gases absorb and/or transmit infrared light at the same wavelength ranges and thus have overlapping adsorption bands (peaks and valleys in the spectrum). Therefore, only a small section of the wavelength range was selected for each gas from which it was identified and quantified. Ideally, the wavelength range chosen for each gas encompasses a spectral peak unique to that gas. However, this was not always possible, especially when multiple wavelength ranges are required to improve the accuracy of gases with a wide concentration range. When there was an overlap in the wavelength range chosen, potential interference from other gases were identified in the software so that the FTIR could account for them in the calculation to properly quantify the gas concentration. The spectral reference concentrations and wavelength ranges, and potential interferences that were selected for each gas were used for all the FTIR data presented in this thesis (Table 2.1).

Table 2.1 Spectral reference concentration, wavelength ranges and potential interferences selected for gases analyzed by the FTIR.

Gas Component	Spectral Reference Concentrations	Wavelength Range (cm⁻¹)	Potential Interferences
H ₂ O	0.1, 0.5, 1, 2, 3, 4, 5, 6 %	3200-3401	CO ₂ , N ₂ O, CH ₄ , NO ₂ , NH ₃
CO ₂	100, 261, 500, 800, 1,000, 1,500, 2,000, 5,000, 10,000, 30,000, 50,000, 75,000 ppmv	926-1204, 2084-2285	H ₂ O, N ₂ O, CH ₄ , NO, NO ₂ , NH ₃
N ₂ O	1, 2, 5, 10.5, 20, 25, 97.5 ppmv	2000-2222, 2540-2590	H ₂ O, CO ₂ , CH ₄ , NO
CH ₄	1, 10, 20, 22.6, 50, 101, 150, 226, 500 ppmv	2700-2980, 3050-3250	H ₂ O, CO ₂ , NO ₂ , NH ₃
NO	2, 5, 10, 20, 50, 100 ppmv	1875-2138	H ₂ O, CO ₂ , N ₂ O, NO ₂
NO ₂	1, 2, 5, 10, 20, 49.5 ppmv	988-1020, 1567-1644, 2550-2600	H ₂ O, CO ₂ , N ₂ O, CH ₄ , NO, NH ₃
NH ₃	5, 20, 206 ppmv	910-1150	H ₂ O, CO ₂ , N ₂ O, NO ₂

The accuracy of the FTIR for identifying and quantifying the concentration of the gases was tested against standard gas concentrations using a flow through testing method. The flow through testing method involved flowing standard gas into the FTIR in-flow port through Teflon tubing at a rate of 2.5 liters per minute (LPM); the flow rate was controlled by a rotameter (0-10 LPM, Cole-Parmer Canada Company, Montreal QC). Another rotameter, used as an off-gassing vent, was connected in line with the tubing using a stainless steel t-connector, positive pressure from this vent ensured that the standard gas was the only source of incoming air. A standard gas was used for each of CO₂, N₂O, CH₄, NO₂, NO, and NH₃, the concentrations for these gases \pm error given by the manufacturer were 10,000 \pm 200, 0.5 \pm 0.01, 10.2 \pm 0.2, 9 \pm 0.18, 13.9 \pm 0.28, and 201.4 \pm 4 ppmv, respectively (Welders Supplies, Winnipeg, MB and Air Liquide, Winnipeg MB). The standard gases that were available for N₂O and CH₄ were in the low concentration range for what was measured from compost windrows for these gases, standard gases for CO₂ and NO

were in the mid concentration range, and NO₂ and NH₃ concentrations were in the high concentration range. Once the concentration of the gas analyzed by the FTIR stabilized the concentration was recorded, then the tubing and FTIR was purged with inert nitrogen gas (N₂). Each standard gas was measured three times and the average was determined. Percent error was calculated for the FTIR versus known standard gas concentrations using the following equation:

$$\text{Percent Error} = \frac{(C_{FTIR} - C_{Std})}{C_{Std}} \times 100 \quad (2)$$

where C_{FTIR} is the concentration of the gas measured by the FTIR (ppmv), and C_{Std} is the known concentration of the standard gas (ppmv).

2.3.2.2 Comparing FTIR Analysis and Gas Chromatography Analysis Over a Range of Gas Concentrations

The accuracy of the FTIR for identifying and quantifying the concentration of gases over a range of gas concentrations was also required to make sure there was no trend in over or underestimating gas fluxes. Since gas fluxes were calculated from the increase in gas concentration over time, if the FTIR randomly over or underestimated gas concentrations, but the rate of increase stayed the same, then the gas flux would not be affected. If, for example, there was a trend in underestimating gas concentrations at the high range but not in the low range this may lead to inaccurate calculation of gas fluxes. To determine the accuracy of the FTIR across a range of gas concentrations, a range of gas concentrations for CO₂, N₂O and CH₄ measured by the FTIR were compared against analysis from a Varian gas chromatograph (GP3800; Varian Canada, Mississauga, ON). To produce a range of gas concentrations to span what could be observed from compost windrows a re-circulating loop was used to conduct the test. To create the re-circulating loop Teflon tubing

connected the FTIR in-flow port to the FTIR out-flow port. A venting port, N₂ gas port and injection port was set up in series along the tubing between the FTIR out-flow and in-flow ports. The venting port and N₂ gas port were rotameters (0-10 LPM, Cole-Parmer Canada Company, Montreal QC), connected with stainless steel t-connectors. The injection port was a stainless steel t-connector with a rubber septum. Inert N₂ gas flowed through the system until the FTIR gas analysis measured zero for all of the gases measured, then the N₂ gas port and venting port were closed simultaneously, creating a re-circulating loop filled with inert gas. Then a needle (100 Microliter Series, Precision Manufacturing Corporation, Baton Rouge, LA, USA) was used to inject the required amount of 25% CO₂, 100% CH₄ or 100% N₂O gas (Welders Supplies, Winnipeg, MB and Air Liquide, Winnipeg MB) to reach the desired concentrations, based on calculations using the total re-circulating loop system volume of 459.216 cm³. The target range for CO₂ gas concentrations were to span from 900 to 60,000 ppmv, the target range for CH₄ was 100 to 550 ppmv, and the target range for N₂O was from 20 to 180 ppmv (based on what the FTIR measured from compost windrows). The FTIR was equipped with an internal off-gassing vent to prevent pressure build-up in the system, adding the gas to the re-circulating loop did not increase the system volume or pressure. After the concentration stabilized, the reading of the FTIR was recorded, then the needle was used to withdraw a 6 mL sample and put it in an evacuated 3 mL Exetainer vial to be analyzed by a Varian GC fitted with an electron capture detector, flame ionization detector, and an automatic sampler (Combi-PAL; CTC Analytics, Zwingen, Switzerland). The concentrations that were measured by the FTIR and GC were plotted on a graph and a linear regression equation and coefficient of determination, the r² value, was calculated to determine if the FTIR and gas chromatograph gave similar gas concentrations.

2.3.2.3 Investigation of Gas Interference

Despite inputting potential interferences in the FTIR software, the re-circulating loop as previously described was also used to check if CO₂ interfered with N₂O at concentrations that could be measured during the composting process. In one test, N₂ gas was run through the loop until all gas concentrations measured were zero, then approximately 0.01 mL of 100% N₂O gas was added, with the expected concentration to measure around 20 ppmv. Then approximately 36 mL of 25% CO₂ gas was slowly added, with the expected concentration to measure around 20,000 ppmv. The final N₂O and CO₂ concentrations were recorded and the change in N₂O concentration was calculated.

In another test, N₂ gas was run through the loop until all gas concentrations were zero. Then approximately 36 mL of CO₂ was quickly added, with the expected concentration to measure around 20,000 ppmv. The result of the N₂O concentration was recorded and the change in N₂O concentration was calculated. The data from these two tests were plotted over time to observe the change in concentrations that occurred.

2.3.2.4 Testing for Ammonia Sorption

It was known that ammonia can be prone to sorption onto surfaces when analyzed and could cause errors in determining NH₃ gas fluxes. To test whether changing the tubing from BeVA to Teflon would reduce NH₃ sorption, a modified version of the flow through testing setup was used. The test also looked at the effect of water vapour to determine if metal surfaces within the system that form condensation were also a potential for NH₃ sorption. The flow through setup had to be modified to allow up to three streams to mix prior to entering the FTIR. A stainless steel cross connector was used to attach three rotameters (0-10 LPM, Cole-Parmer Canada Company,

Montreal QC), for flow of NH₃ or CO₂ standard gases, water vapour, and N₂ to allow mixing of gas prior to flowing through a 12 m length of BevA or Teflon tubing into the FTIR in-flow port. The CO₂ gas was used as the control gas to compare against the behaviour of NH₃. Total flow rate going in was monitored to match the flow rate going out by placing additional rotameters just before and after the FTIR in-flow and out-flow ports, respectively. The NH₃ or CO₂ gas was set to flow at a rate of 1 LPM. Water vapour was produced using a LI-610 Portable Dew Point Generator (LICOR BioSciences Inc., Lincoln, NB), and flowed in at a rate of 1.5 LPM; there was no condensation observed on the surfaces of the system. Inert N₂ gas was used to top up the flow rate to 2.5 LPM if needed, as well as purge the system. The mixing time, purge time, and steady state gas concentration were recorded. The mixing time referred to the length of time it took for the gas being tested to reach a steady state concentration after it was turned on. The purge time referred to the length of time it took for the gas to purge from the system after the CO₂ or NH₃ and water vapour was turned off and N₂ was turned up to 2.5 LPM. The steady state concentration was the concentration, in ppmv, of NH₃ gas measured by the FTIR. Each variation of the tests (example CO₂ or NH₃ gas, with or without water, using BevA or Teflon tubing) was conducted three times. Statistical analyses were performed with SigmaPlot Systat Software (SigmaPlot for Windows Version 11.0, Germany). A two-way analysis of variance (ANOVA) was used to determine if there were differences in the behaviour (mixing time and purging time) between CO₂ and NH₃ and the effect of tubing (Bev A or Teflon). A Fisher Least Significant Difference (LSD) test was used to determine significant differences ($P < 0.05$). Data was tested for normality and equal variance (Shapiro-Wilks). A two-way ANOVA was also used to determine if there were differences in the behaviour (mixing time, steady state concentration, and purging time) of NH₃ with and without water vapour added and the effect of tubing (Bev A or Teflon). Again, a Fisher LSD test was used

to determine significant differences ($P < 0.05$) and data was tested for normality (Shapiro-Wilks) and equal variance.

Another test of NH_3 sorption was conducted in the lab using a modified version of the automated chamber and FTIR system to simulate field conditions and increase the potential for sorption to surfaces or condensation. A stainless steel t-connector was used to attach two rotameters (0-10 LPM, Cole-Parmer Canada Company, Montreal QC) that controlled the flow rate of NH_3 or CO_2 standard gases, and water vapour to allow mixing of gas prior to flowing through a hole drilled into the side of the PVC collar. The bottom of the PVC collar was sealed to the base of the table using Vaseline. An automated chamber with either BeVA or Teflon tubing was fitted over the PVC collar and a 6 m length of corresponding BeVA or Teflon extension tubing ran the gas to the multiplexer and then BeVA or Teflon tubing to the FTIR in-flow port and from the FTIR out-flow port back to the multiplexer, and finally another 6 m length of BeVA or Teflon tubing back to the chamber. In the scenario with Teflon tubing, the inside of the chamber dome was also covered with a Teflon adhesive tape (PTFE Tape Coated Glass Fabric with Adhesive Backing, Johnston Plastics, Winnipeg, Manitoba). The multiplexer was also re-fitted to have BeVA tubing instead of urethane tubing where it was accessible. Teflon tubing was not available in the required sizes to redo the tubing inside the multiplexer. During the tests the chamber was closed for 3 minutes. The accumulation time, purge time, and peak concentration were recorded. The accumulation time referred to the time between the chamber dome closing until the concentration of the gas reached its maximum. The purge time referred to the length of time it took for the gas to purge from the system back to baseline values once the chamber dome had lifted. The peak concentration referred to the concentration of the NH_3 gas at the maximum concentration during the accumulation time in ppmv. Each variation of the test (example CO_2 or NH_3 gas, with or

without water, using BevA or Teflon tubing) was conducted three times. Statistical analyses were performed with SigmaPlot Systat Software (SigmaPlot for Windows Version 11.0, Germany). A two-way ANOVA was used to determine if there were differences in the behaviour (accumulation time and purging time) between CO₂ and NH₃ and the effect of tubing (Bev A or Teflon). A Fisher LSD test was used to determine significant differences ($P < 0.05$). Data was tested for normality (Shapiro-Wilks) and equal variance. A two-way ANOVA was also used to determine if there were differences in the behaviour (accumulation time, peak concentration, and purging time) of NH₃ with and without water vapour added and the effect of tubing (Bev A or Teflon). Again, a Fisher LSD test was used to determine significant differences ($P < 0.05$) and data was tested for normality (Shapiro-Wilks) and equal variance.

2.3.3 Normalizing Gas Concentrations to Assess Gas Accumulation Behavior

It was difficult to simulate gas emissions from compost windrows in lab analysis, therefore, to assess the accumulation behaviour of the gases, measurement periods during the measurement of actual field gas fluxes from Chapter 5 were examined. The behaviour of CO₂, CH₄, N₂O and NO gases was assessed from six measurements from a single chamber, on November 9, 2012, as all gases were being emitted simultaneously; the compost windrow was setup just three weeks prior, on October 15, 2012, and was in the active stage but was anaerobic. Fluxes of NH₃ were not measured on November 9, 2012, but there were fluxes measured earlier on November 2, 2012. The NH₃ fluxes exhibited different behaviours amongst four chambers that were being used on that day, so the behaviour during six measurement periods from each of the four chambers were assessed. To assess the behaviours of the gases the concentrations were normalized using a 0 to 1 scale during the entire measurement period, including a 17-minute pre-purge, 3-minute

observation, and 10-minute post-purge time, for a total of 30 minutes. Normalization was calculated using the following equation:

$$S = \frac{(C - C_{min})}{(C_{max} - C_{min})} \quad (3)$$

where S was the normalized value, from 1 to 0, C was the original concentration (ppmv), C_{min} was the minimum concentration in the measurement period (ppmv), and C_{max} was the maximum concentration in the measurement period (ppmv). The normalized values were used to compare the time at which the concentration started to accumulate, the total time the concentration was accumulating, the slope of the rise in concentration during the accumulation time, and the r^2 value during the accumulation time. The time at which the gas concentration accumulation started was determined when the normalization value was > 0.05 , this indicated there was an increase in concentration compared to the background. Total accumulation time was the time between the gas concentration accumulation start time and the peak time, when the standardization value equalled 1. The slope and r^2 were calculated using the values during the total accumulation time. Carbon dioxide was used as the comparison gas because it was always emitted throughout the composting process; in order to use the Excel Macro a gas that was consistently emitted was required to set the time at which to select the linear accumulation period, used for calculating gas fluxes. If N_2O , CH_4 , and NO gases behaved the same as CO_2 then the Excel Macro can be used for them as well.

2.4 Results and Discussion

2.4.1 Verification of Gas Concentrations Measured by the FTIR

The FTIR was able to correctly identify the gases present in each of the standard gases. Further, when quantifying the gas concentration, the FTIR measurements for CO₂, N₂O, and CH₄ were within the \pm concentration error given by the manufacturer of the standard gases; however, NO, NO₂, and NH₃ gases were overestimated by the FTIR outside the standard error given by the manufacturer (Table 2.2). The NO, NO₂, and NH₃ gases could be more reactive compared to CO₂, N₂O, and CH₄, therefore the FTIR was not as accurate at measuring them. The standard error of the NO, NO₂, and NH₃ gases measured by the FTIR was low which indicated that the FTIR was consistent in quantifying the concentration. If the FTIR consistently overestimates these concentrations, the rate of concentration accumulation over time, used for flux calculation may not affect the flux rate.

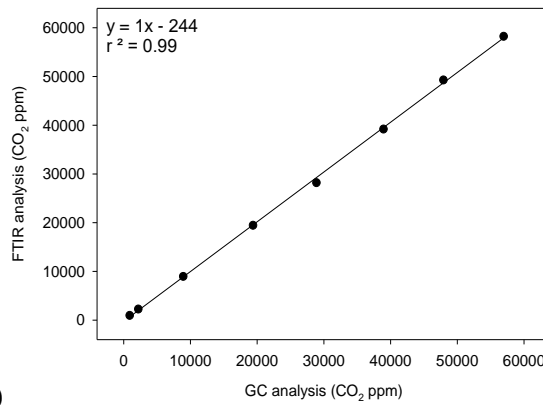
Table 2.2 FTIR measurements of standard gases using the flow through testing technique and percent error compared to given standard gas concentrations.

Gas	Standard Gas (ppmv)	FTIR Measurement (ppmv)	Percent Error (%)
CO ₂	10,000 \pm 200	10,196.84 (14.99)	2.0
N ₂ O	0.5 \pm 0.01	0.503 (0.02)	0.6
CH ₄	10.2 \pm 0.2	10.03 (0.14)	-1.7
NO	13.9 \pm 0.28	15.51 (0.25)	11.6
NO ₂	9 \pm 0.18	11.37 (0.04)	26.3
NH ₃	201.4 \pm 4	215.69 (1.08)	7.1

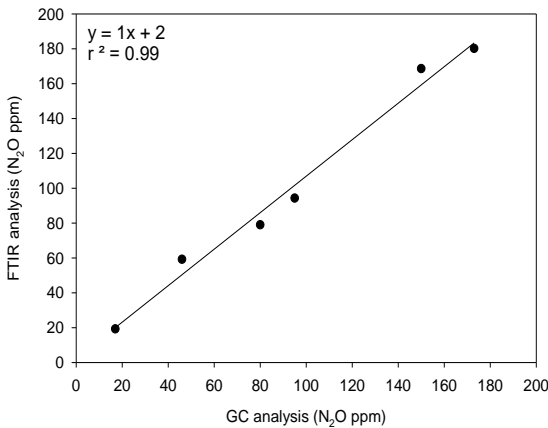
The standard gas included values \pm range of error given by the manufacturer; FTIR measurement includes values with the mean of three samples ± 1 standard error in parenthesis. Percent error is calculated assuming the standard gas is the actual concentration.

The comparison between the FTIR and GC measurements for CO₂, N₂O, and CH₄ appears to have a linear fit over the range of concentrations (Figure 2.3). The high coefficients of determination, r^2 values, suggest a linear model explained 99% of the response variable variation

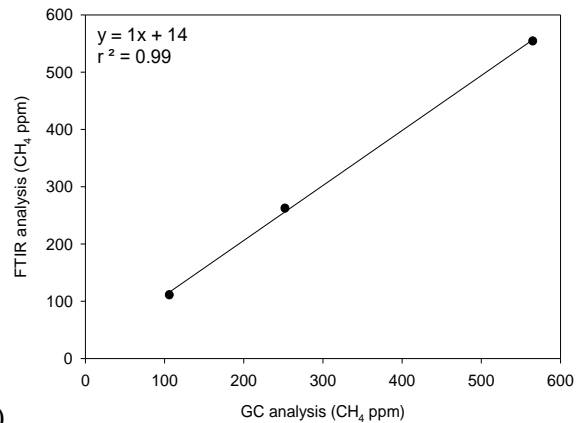
(Figure 2.3). The data points were randomly above or below the linear model, suggesting random variation. The FTIR was capable of producing measurements similar to the GC across these wide ranges of gas concentrations. There was no trend in over or underestimating gas concentrations at high or low concentration ranges, therefore the rate of gas fluxes should not be affected. It would be ideal to analyze NO, NO₂, and NH₃ gases across a range of concentrations expected from compost windrows. However, since the GC cannot measure NO, NO₂, and NH₃ multiple standard gas concentrations would be required for the test.



a)



b)



c)

Figure 2.3 Comparison of FTIR and GC measured concentration values of CO₂ (a), N₂O (b), and CH₄ (c) gases including linear fit equation ($y=mx+b$) and r^2 value.

2.4.2 Analysis of Gas Interference

In the first gas interference test, a re-circulating loop contained inert N₂ air and all gas concentrations were at 0 ppmv. After N₂O was added, the N₂O concentration increased to 26.5 ppmv (Figure 2.4). Then CO₂ was added slowly which resulted in a final CO₂ concentration of 18,000 ppmv and N₂O concentration of 24 ppmv (Figure 2.4). The N₂O concentration dropped 2.5 ppmv or a 10 % decrease in concentration. There was no observable change in any of the other gases during this test (data not shown).

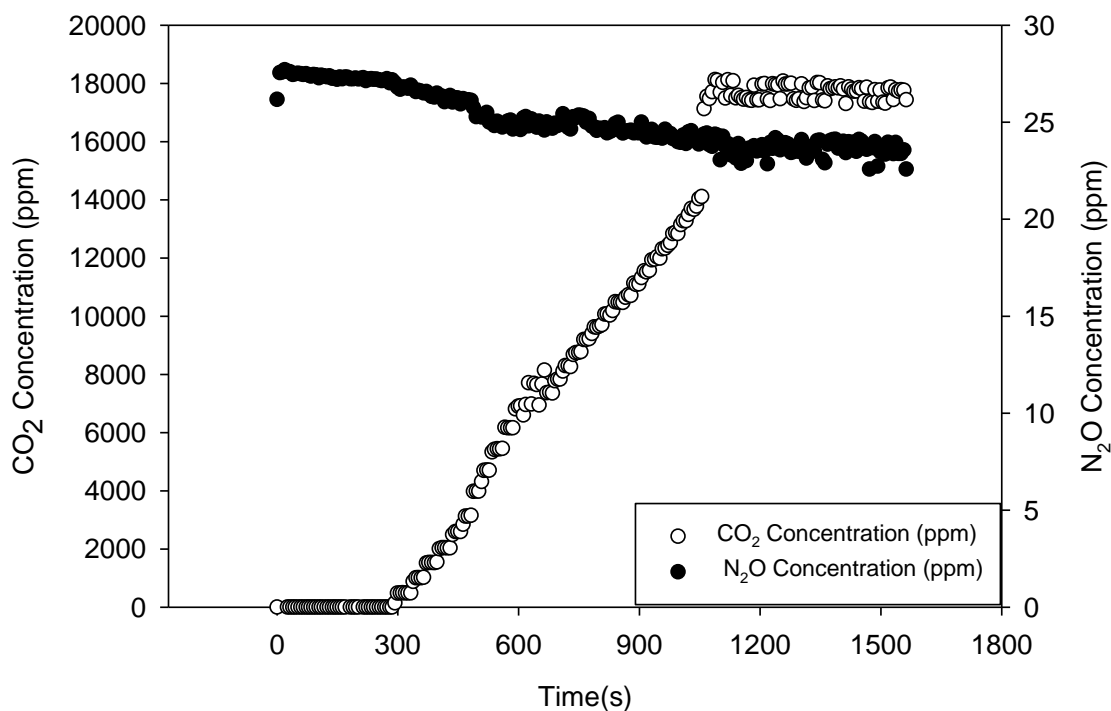


Figure 2.4 Interaction of CO₂ and N₂O when CO₂ was added to air with a 26.5 ppmv concentration of N₂O using the re-circulating loop technique.

In the second gas interference test, a re-circulating loop contained inert N₂ air and all gas concentrations were at 0 ppmv. After CO₂ was quickly added the CO₂ concentration increased to 20,000 ppmv and the N₂O concentration decreased to -0.5 ppmv (Figure 2.5). There was no observable change in any of the other gases during this test (data not shown). The FTIR settings

that were chosen limited error from gas interferences between most of the gases; however, these two tests suggest there was an interference that caused N₂O to decrease when CO₂ increased. It was possible to change the FTIR settings and re-process the data, however, selecting different FTIR settings, reference spectra, wavelength ranges, or interferences did not noticeably reduce the effect of interference of CO₂ on N₂O any better than what was shown here.

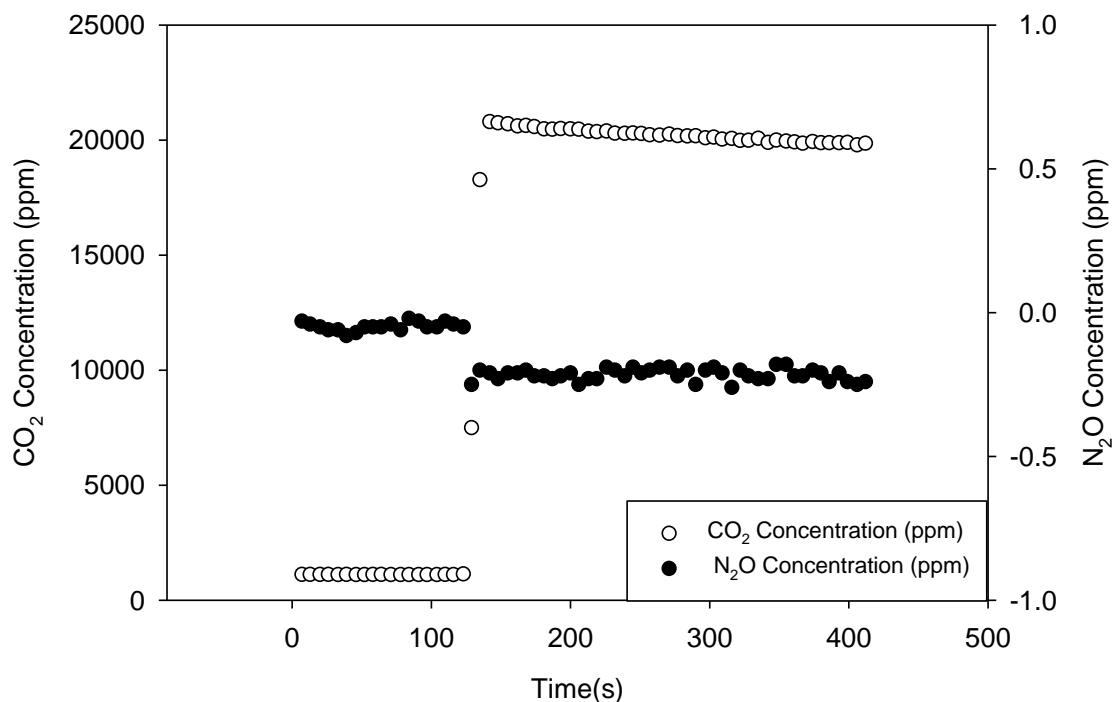


Figure 2.5 Interaction of CO₂ and N₂O when CO₂ was added to air with a zero concentration of N₂O using the re-circulating loop technique.

Another potential reason for N₂O to decrease when CO₂ was added could have been due to dilution of N₂O gas. The addition of 2 % CO₂ concentration could result in the dilution of the N₂O gas by 2 %. Additionally, the FTIR had an internal off-gassing vent to maintain the pressure of the system (Gasmeter Technologies, 2009). Injecting 36 mL of gas to the system may have displaced air in the re-circulating loop that may have not contained a proportionate amount of N₂O and CO₂.

The amount that displacing the air would decrease N₂O by cannot be determined; therefore, it was unknown if these factors would account for the perceived interference.

Since the amount N₂O decreased by when CO₂ was added differed depended on the starting concentration of N₂O it would not be practical to determine a correction factor because of the variation in CO₂ and N₂O concentrations measured from compost windrows. Further investigation could be required especially if analysis was for sensitive data. Since compost windrows had high fluxes of N₂O the degree to which CO₂ had an effect on N₂O may not be a relatively important source of error compared to research done on soils with generally lower N₂O fluxes.

2.4.3 Analysis of Ammonia Sorption

The flow through testing method determined that there was no statistical difference between gases, CO₂ and NH₃, for the mixing time, but there was a statistical difference in the purge time (Table 2.3); NH₃ took longer to purge from the system compared to CO₂. This result indicated there was potential for ammonia sorption. There was no statistical difference between tubing type, BevA or Teflon, for mixing time or purge time (Table 2.3). There was also no interaction between the type of gas and the type of tubing. The flow through testing method also determined that there was no statistical difference between NH₃ with water and NH₃ without water, for the mixing time and purging time, but there was a statistical difference in the steady state concentration (Table 2.3); the NH₃ with water had a higher steady state concentration. The higher concentration when water was present was unexpected; water in the system was expected to adsorb NH₃ preventing NH₃ from reaching the sample cell (Harper, 2005). There was no statistical difference between tubing type, BevA or Teflon, for mixing time, steady state concentration, or purge time (Table 2.3). There was also no interaction between water vapour and tubing type.

Table 2.3 Comparison of the type of gas and effect of tubing on mixing time(s) and purge time (s), as well as the comparison of NH₃ with or without water and the effect of tubing on mixing time (s), steady state concentration (ppmv) and purge time (s) using the flow through testing technique.

	Mixing Time (seconds)	Steady State Concentration (ppmv)	Purge Time (seconds)
Gas			
CO ₂	53(3)		98(5)*
NH ₃	58(2)		258(12)*
Tubing			
Bev A	57(3)		175(37)
Teflon	53 (2)		181(37)
Water vapour			
NH ₃ without water vapour	58(2)	100(1)*	258(29)
NH ₃ with water vapour	60(4)	110(1)*	252(4)
Tubing			
NH ₃ in Bev A	58(2)	104(2)	255(11)
NH ₃ in Teflon	60(4)	105(3)	256(7)

Values for accumulation time, steady state concentration and purge time are the mean of three tests, ± 1 standard error in parenthesis. Asterisk (*) in the same grouping in the same column were significantly different at $P < 0.05$ by two-way ANOVA Fisher LSD.

The re-circulating loop was used to stimulate actual conditions of the automated chamber and FTIR system in use. This test determined that there was no statistical difference between gases, CO₂ and NH₃, for the accumulation time, but there was a statistical difference in the purge time; again NH₃ took longer to purge from the system compared to CO₂ (Table 2.4). This result again indicated there was potential for ammonia sorption. There was no statistical difference between tubing type, BevA or Teflon, for accumulation time or purge time (Table 2.4). There was also no interaction between the type of gas and the type of tubing. The re-circulating loop testing method also determined that there was no statistical difference between NH₃ with water and NH₃ without water, for the accumulation time and purging time, but there was a statistical difference in the steady state concentration (Table 2.4); the NH₃ without water had a higher steady state concentration. This is different compared to the result of the flow through testing method and

indicated, as expected, that water in the system can adsorb NH_3 , and prevent NH_3 from reaching the sample cell (Harper, 2005). There was no statistical difference between tubing type, BevA or Teflon, for accumulation time, steady state concentration, or purge time (Table 2.4). There was also no interaction between water vapour and tubing type.

These tests indicated that changing the tubing to Teflon instead of BevA would not make a difference in reducing the potential sorption of NH_3 ; therefore, it was decided to keep the extension tubing as BevA. The flexibility of BevA compared to Teflon can avoid kinking the tubing when it is moved in the field and avoid damage to the tubing that could cause leaks. Tubing that connected the multiplexer, Li8100 IRGA analyzer and FTIR was changed to Teflon tubing because this tubing did not get moved very often.

Although actual condensation was not observed with the water vapour addition, there were differences in the steady state concentration of NH_3 with and without water. There were also differences in the purge time of CO_2 and NH_3 . These observations indicate that there was potential for NH_3 sorption in the condensation that could be formed on metal surfaces such as the aluminum manifold of the multiplexer or the chamber dome, as well as the extension tubing. From these tests it was not possible to determine which component NH_3 was adsorbing to or how to prevent it. The aluminum manifold of the multiplexer was currently not available in a material designed to reduce water condensation. The multiplexer was kept in an insulated and heated shelter when used in the field to prevent condensation on the aluminum manifold. It was impractical to heat the extension tubing or chamber domes to reduce condensation due to the energy requirement and to maintain the functionality of the chamber domes. After testing was completed, a personal communication with Jason Hupp (2012), Sr. Applications Analyst, LI-COR Biosciences suggested NH_3 could also get trapped on the Balston filters (filters that filter air exiting the multiplexer), if the filters were

not changed frequently. Testing was not done to determine if this was a factor, however the filters were changed every week when NH₃ was emitted for the study conducted in Chapter 4.

Table 2.4 Comparison of the type of gas and effect of tubing on mixing time(s) and purge time (s), as well as the comparison of NH₃ with or without water and the effect of tubing on mixing time (s), steady state concentration (ppmv) and purge time (s) using the re-circulating loop testing technique.

	Accumulation Time (seconds)	Steady State Concentration (ppmv)	Purge Time (seconds)
Gas			
CO ₂	193(1)		369(13)*
NH ₃	191(1)		820(28)*
Tubing			
Bev A	192(1)		602(101)
Teflon	191(1)		587(105)
Water vapour			
NH ₃ without water vapour	191(1)	130(2)*	820(28)
NH ₃ with water vapour	195(2)	97(2)*	804(19)
Tubing			
NH ₃ in Bev A	194(2)	112(8)	803(30)
NH ₃ in Teflon	192(2)	115(7)	820(15)

Values for accumulation time, steady state concentration and purge time are the mean of three tests, ± 1 standard error in parenthesis. Asterisk (*) in the same grouping in the same column were significantly different at $P < 0.05$ by two-way ANOVA Fisher LSD.

2.4.4 Analysis of Gas Behaviour Measured from Compost Windrows

Normalized gas concentrations of CO₂, N₂O, and CH₄ for a chamber on November 9, 2012 show that these gases, when emitted, behave fairly similar (Figure 2.7). After the pre-purge began the gases remained at background levels, once the chamber dome closed there was a slight pause to allow the gases to mix before the concentration started to accumulate. Once the chamber opened the gas concentration immediately started to decline as the gases were purged from the system, when the post-purge period ended the multiplexer switched to a different chamber. The slope during the accumulation period was also similar for these gases and they had a high r^2 value

indicating the variability could be explained by a linear model (Table 2.5). The NO had a similar pattern to the other gases; however, the time the gas started to accumulate was delayed (Figure 2.6). The NO gas concentration immediately declined when the chamber opened as with the other gases (Figure 2.6). The slope of NO during the accumulation time was steeper than the other gases because the total accumulation time was shorter (Table 2.5). The r^2 value was a bit less than the r^2 values of the other gases but a linear model would still be considered a good fit (Table 2.5), the increased variability amongst the residuals, could have been because NO was a more reactive gas and the FTIR was less accurate at measuring NO concentration, as demonstrated previously (Section 2.4.1).

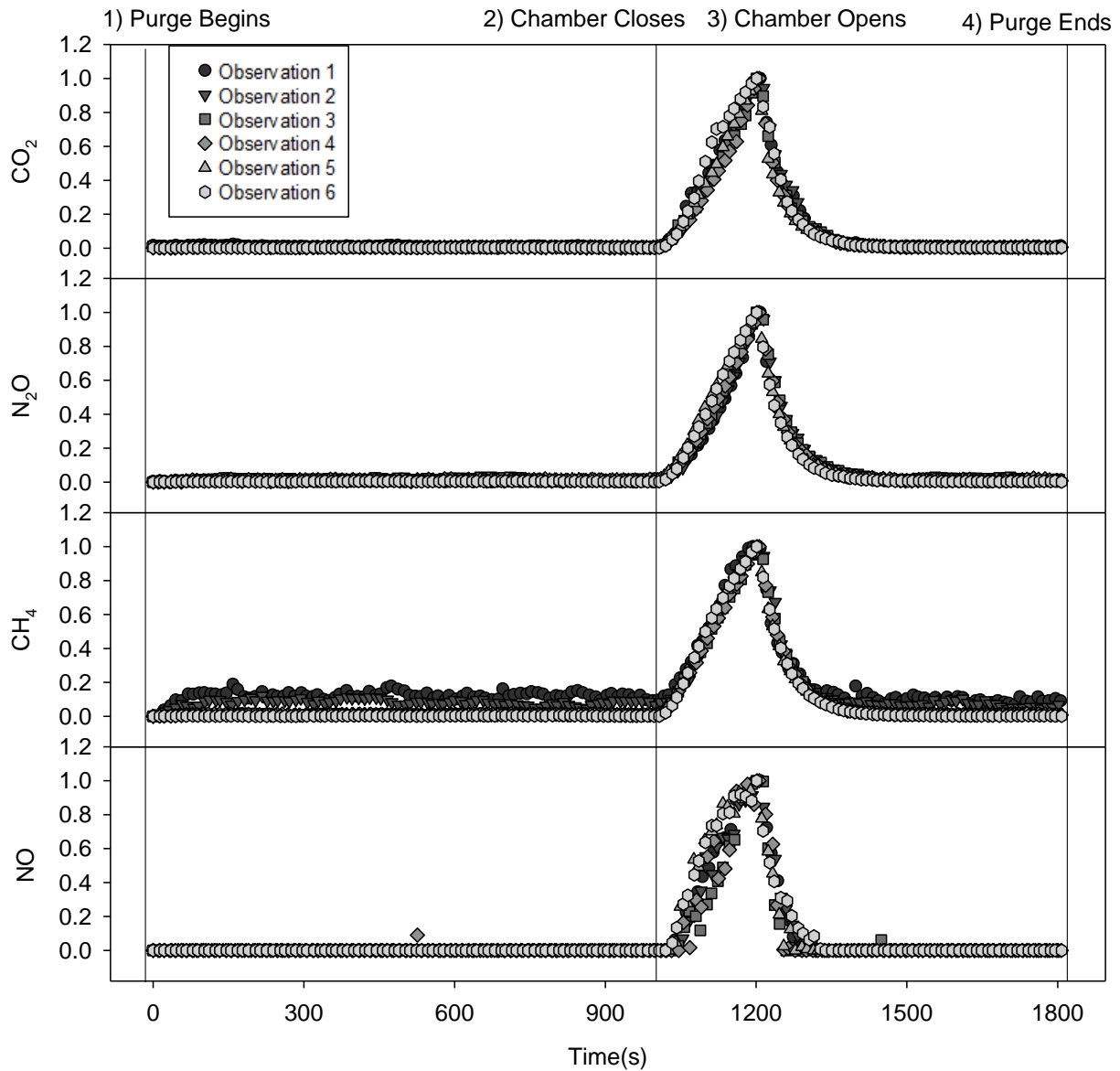


Figure 2.6 Normalized concentration values of CO₂, N₂O, CH₄, and NO gases versus time (seconds) for six chamber observations from a chamber on November 9, 2012. The vertical lines indicate the start of the pre-purge, closing of the chamber, opening of the chamber, and the end of the post-purge periods, respectively.

Table 2.5 Summary of CO₂, N₂O, CH₄, and NO gas behaviour from normalized values of six chamber observations from a chamber on November 9, 2012.

Gas	Total Pre-Purge Until Accumulation Time (seconds)	Total Gas Accumulation Time (seconds)	Slope	r ²
CO ₂	1,019(1)	184(2)	0.0055(0.0001)	0.98(0.00)
N ₂ O	1,023(3)	180(3)	0.0055 (0.0001)	0.98(0.01)
CH ₄	1,021(2)	180(2)	0.0056 (0.0001)	0.99(0.00)
NO	1,041(4)	163(3)	0.0059 (0.0002)	0.92(0.02)

Values for total pre-purge until accumulation time, total gas accumulation time, slope, and r² are the mean of six observations ±1 standard error in parenthesis.

Normalized gas concentrations of NH₃ gas from Chambers 1, 2, 3 and 4 on November 2, 2012 indicated four different scenarios occurred when NH₃ was emitted (Figure 2.7). For Chamber 3, after the automated chamber dome closed there was a steady increase in NH₃ concentration, when the chamber dome opened the concentration continued to increase at approximately the same rate for about 20 seconds and then gradually declined (Figure 2.7). The scenario in Chamber 3 was the most common observed scenario exhibited when NH₃ was emitted on other days from other chambers. For Chamber 4, there was a steady increase in NH₃ concentration then immediately after the automated chamber dome opened there was a rapid spike in NH₃ concentration that increased for about 45 seconds and then gradually declined (Figure 2.7). This scenario occurred when the NH₃ concentrations being measured reached relatively high values. For Chamber 1, there was still a small amount of residual NH₃ that was adsorbed to the system and only now purging from the tubing, when the chamber closed (Figure 2.7). The circulating air had a lower NH₃ concentration than when the automated chamber dome was open, so the concentration appeared to decline until the automated chamber dome opened again. The scenario of Chamber 2 was very similar to Chamber 1 except when the automated chamber dome was closed the concentration

remained the same as if the automated chamber dome was open; there was no change in NH_3 concentration (Figure 2.7). These last two scenarios only occurred when the NH_3 concentration were relatively lower.

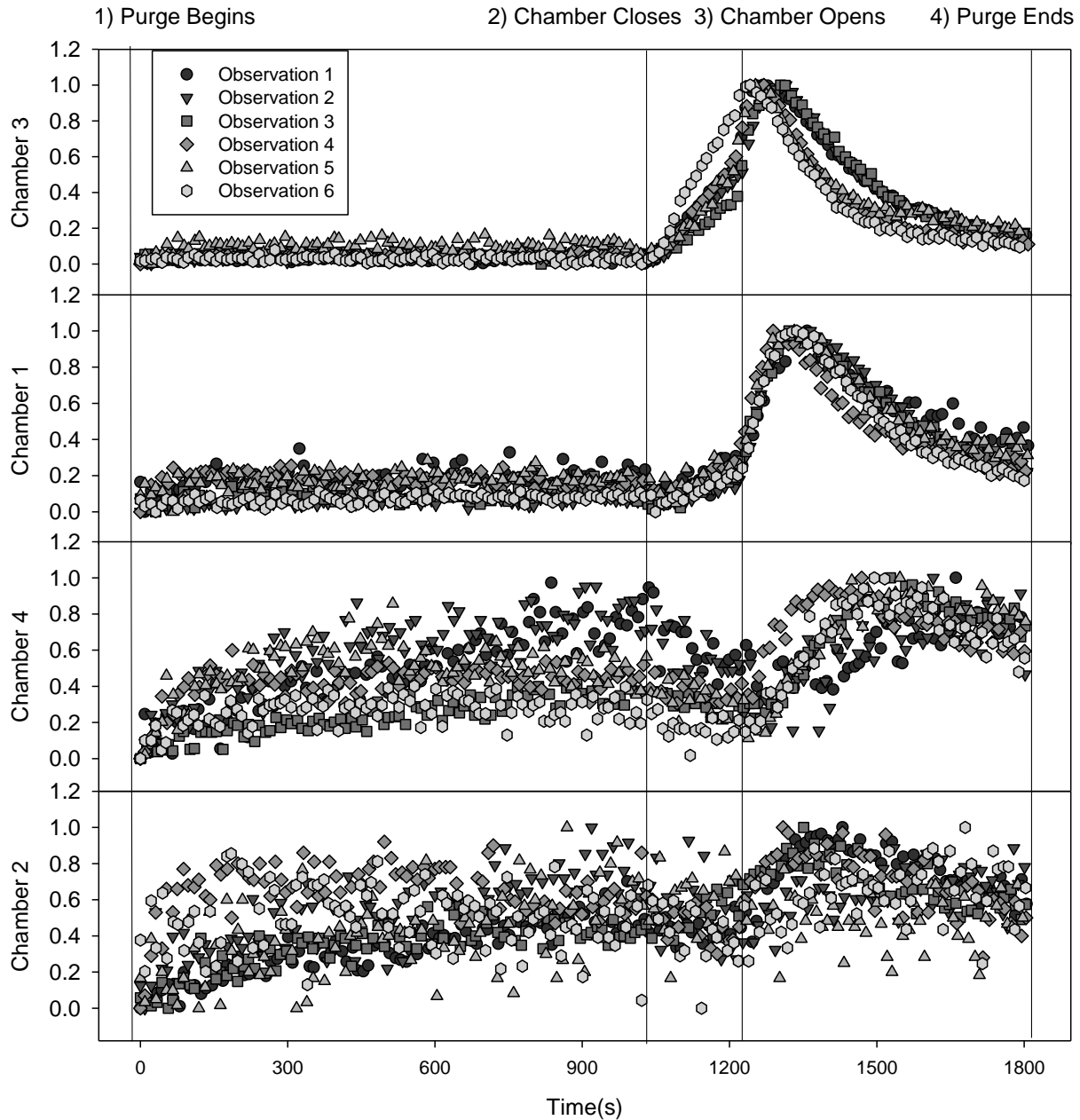


Figure 2.7 Normalized concentration values of NH_3 gas versus time (seconds) for six observation periods from Chambers 1, 2, 3, and 4 on November 2, 2012. The lines indicate the start of the pre-purge, closing of the chamber, opening of the chamber, and the end of the pre-purge, respectively.

Selection of an appropriate linear portion of the accumulation period was necessary for an accurate estimation of gas flux. For ease of data processing it was ideal to have a method for selecting a calculation period that consistently gives reliable results for all the gases. Gases including CO₂, CH₄ and N₂O were shown to behave similarly, thus the use of the Excel Macro that was developed to select the linear accumulation period should give a reliable estimation of gas flux for these gases. However, relying on the Excel Macro to select the accumulation period for NO may result in selection of a time when the accumulation period was not linear, causing under or overestimation of the gas flux. Analysis of NO fluxes from additional field collection measurements from compost windrows indicated that the higher the peak NO concentration was, then the earlier NO started to accumulate. Therefore, there could potentially be an issue with the minimum detection level of NO resulting in a delayed response of NO concentration. It was decided to manually adjust the time period used for flux calculations if the Excel Macro selected a time that resulted in the r^2 value to be less than 0.5 when NO was known to be emitted. Usually this only required shifting the time period one to three data points or ten to thirty seconds. This was possible for NO because it was not emitted that frequently.

For NH₃ it was decided to use the Excel Macro and the same selection process for selecting the linear accumulation period as CO₂, CH₄, and N₂O, knowing that it may result in miscalculation of gas flux. For the scenario exhibited by Chamber 3, the linear accumulation period selected by the Excel Macro should give a reliable estimation to calculate a gas flux, which was beneficial since this was the most frequent observation scenario. For the scenario exhibited by Chamber 4 it was likely that the NH₃ was trapped in condensation on surfaces within the system when the chamber was closed, and then evaporated when the chamber dome opened releasing NH₃; therefore, the linear accumulation period chosen could underestimate gas flux. In future studies, if

this scenario is exhibited again, area under the curve analysis could be estimated and possibly be used to calculate NH₃ fluxes. The scenario exhibited by Chamber 1 resulted in the selection of a time period that showed a negative NH₃ gas flux. Since the concentrations were low during this time this should result in little impact on the overall NH₃ flux so the data was not altered; however, in future studies if this scenario was exhibited again there would be justification to change the value to neutral (zero) NH₃ flux or remove the data point. The scenario exhibited by Chamber 2 will result in selection of a time period where the gas flux would show random variability, some would be slightly positive, some would be slightly negative, it was assumed that the values would cancel each other out there was no need to alter or remove the data. The post-purge time should have been extended to fully flush out the residual NH₃.

2.5 Conclusion

This was the first attempt to combine an automated chamber system in series with a FTIR analyzer for measuring gas emissions from windrows during the composting process. Conducting laboratory tests and analyzing actual flux data from compost windrows provided a better understanding of the standard operating procedures required and the potential challenges of using this system. The FTIR proved to be capable of measuring CO₂, CH₄, and N₂O gas concentrations accurately including across a range of concentrations that could be seen when measuring gas fluxes from compost windrows. The FTIR was not able to measure NH₃, NO, and NO₂ within the error given by manufacturers of standard gases, possibly because these gases were more reactive. However, the FTIR was consistent in over estimating these concentrations, therefore the over estimated gases may not necessarily result in poor estimation of gas fluxes. The FTIR settings take

into account possible interferences; however, an underestimation of N₂O when CO₂ was present was observed. The underestimation of N₂O could have been due to interference, or other possibilities such as dilution or displacement of air when using the re-circulating loop to do the testing because of the amount of CO₂ that was required to be added to reach concentrations observed when measuring gas fluxes from compost windrows. The potential interference of N₂O was acknowledged, but due to the relatively high fluxes of N₂O from compost windrows, the error may not be relatively important compared to more sensitive applications such as measurement of N₂O fluxes from soil.

The potential sorption of NH₃ was observed by analyzing NH₃ fluxes from actual compost windrows as well as in laboratory analysis. The laboratory testing confirmed that NH₃ took longer to purge compared to CO₂. There were also differences in the steady state concentration of NH₃ with or without water vapour present. To reduce the potential for condensation the multiplexer was kept in a heated and insulated storage container when doing field measurements to minimize condensation on the aluminum manifold. Heating the chamber domes or extension tubing seemed impractical due to the energy requirements and functionality of the chambers. After the testing was conducted, the Balston filters, that are used to filter air exiting the multiplexer, were suggested as another potential source of NH₃ sorption and it was recommended to change the Balston filters once per week when NH₃ was emitted. There was no difference or reduction in NH₃ sorption when either BeVA tubing or Teflon tubing was used; therefore, BeVA tubing continued to be used as extension tubing due to the flexibility of that type of tubing.

Analysis of actual observation data measured using the automated chamber and FTIR system on compost windrows indicated similar behaviour of CO₂, CH₄, and N₂O gases. The Excel Macro that was developed should be able to automatically select data points for use in gas flux

calculations. The selection of data points for NH_3 and NO fluxes had to be checked manually for accuracy due to inconsistent linear accumulation in concentration over time. Analysis of actual NH_3 fluxes indicated there could be different scenarios depending on the amount of NH_3 being emitted and the sorption that was occurring. Different techniques for calculating gas fluxes, such as estimating the area under the curve, could be used in future studies. Additionally, the post-purge time could also be extended to allow residual NH_3 to fully purge from the system.

Overall the automated chamber system and FTIR can be used to measure gaseous emissions during the composting process, there is potential to utilize this technology to develop best management practices for reducing greenhouse and nitrogen gaseous losses. The automated chamber and FTIR system allowed data collected to be collected every half hour, which will be advantageous over static vented chambers as that frequency of data collection can allow understanding of the rapidly changing conditions and dynamics of gas emissions during the composting process. Collecting frequent data reduces the need to gap fill between fluxes and could increase confidence in the results. The ability to measure NH_3 and NO , despite the limitations, in addition to N_2O can give an indication of composting practices that are prone to N losses. Additionally, this system could be used to advance the understanding of the microbiology activity during composting, especially in relation to the different types of N gases produced and the factors involved in producing one type of N gas over another. This area of research could be of interest to those that are manufacturing very high quality composts.

3.0 SUITABILITY OF COMPOSTING SEPARATED SLURRY SOLIDS THROUGHOUT THE WINTER AND MEASUREMENT OF GREENHOUSE GAS AND NITROGEN GASEOUS LOSSES

3.1 Abstract

The suitability of using composting throughout the winter as an on-farm management tool for separated pig slurry solids and straw was assessed to determine if thermophilic temperatures could be reached to meet the Process to Further Reduce Pathogens (PFRP) criteria and produce a mature finished product with consistent agronomic parameters. There were three windrows, one started in each of September (Trial 1), October (Trial 2), and November (Trial 3). The quantity and timing of greenhouse gas and nitrogen gaseous losses, including carbon dioxide (CO₂), methane (CH₄), nitrous oxide (N₂O), ammonia (NH₃), nitric oxide (NO), and nitrogen dioxide (NO₂), were also determined for the first two Trials using an automated chamber system in-line with a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer to determine if it was capable of relating emissions to physical properties and management of the windrows. Emissions of CH₄ and N₂O were higher in Trial 2 than in Trial 1, at 97.9 kg CO₂-equivalent Mg⁻¹ (initial dry mass) and 46.5 kg CO₂-equivalent Mg⁻¹, respectively, due to higher moisture content at the start of Trial 2. In terms of N gaseous emissions, there was very little difference between Trial 1 and 2 with a total of 0.22 g N Mg⁻¹ (N₂O 0.10 g N Mg⁻¹, NO 0.02 g N Mg⁻¹, and NH₃ 0.10 g Mg⁻¹) and 0.26 g N Mg⁻¹ (N₂O 0.10 g N Mg⁻¹, NO 0.01 g N Mg⁻¹, and NH₃ 0.15 g Mg⁻¹), respectively. Total losses of C, 396 kg Mg⁻¹ and 367 kg Mg⁻¹, and total N, 12.9 kg Mg⁻¹ and 12.5 kg Mg⁻¹, calculated by mass balance, for Trial 1 and 2 respectively, were not accounted for by the gaseous emissions measured. There was potential that the chambers did not capture fluxes of CH₄ and NH₃, but considerable

losses could also be from leaching, runoff, and removal of material during sampling and turning events as well. All windrows were able to reach thermophilic temperatures in order to meet PRFP criteria; however, windrow turning at the start of the composting process needed to be frequent enough to meet the turning requirements of the PRFP criteria as well. The compost produced was considered a mature and consistent product ready to be applied in the spring despite visible pieces of straw still present in the final product.

3.2 Introduction

Over the years repeated application of pig slurry in the RM's of La Broquerie and Hanover has built a surplus of soil nutrients, particularly phosphorus (P), in soils close to barns (Statistics Canada, 2006). This practice poses an environmental risk to surface water when the excess P runs off into waterways and can cause eutrophication in lakes (Kumaragamage et al., 2009). In an effort to reduce this impact, the regulations in Manitoba have changed, as of November 2013, manure application rates can only be applied based on soil P instead of soil nitrogen (N), reducing the amount of pig slurry that can be spread onto the same soil year after year. Many pig production facilities in Manitoba have earthen, concrete or steel structures to store pig slurry manure until it can be applied to fields, and many of these are already at capacity because the slurry can only be applied in the spring or fall (Agricultural Guidelines Development Committee. 2007). Building more facilities to store manure can be expensive and does not provide a solution for manure distribution in the long term. There are also many deterrents to marketing pig slurry including the cost to transport the pig slurry exceeding its value because the water content dilutes the concentration of nutrients, the odorous process of storing and applying the slurry and the challenge of distributing nutrients throughout the slurry and uniformly applying them across the

field. In response, pig producers are looking for options that provide economic solutions to distribute the nutrients to P deficient soils. The Prairie Agricultural Machinery Institute (PAMI) and its collaborators completed a comprehensive evaluation on the suitability of centrifugation of pig slurry using a two-phase clarifying decanter (ALSYS 20, Alfa Laval Inc. Scarborough, ON). Centrifuging the pig slurry into liquid and solid fractions can concentrate 63 % of the total labile P in the solid fraction allowing continued use of the liquid fraction to be applied to soil nearby (Kumaragamage et al. 2014). Once the liquid fraction was removed, the separated pig slurry solids were lighter and had a higher concentration of nutrients per volume; therefore, the cost to transport further distances from the point of generation was reduced. In addition, much of the odour generated by storing and applying pig slurry was due to the anaerobic decomposition of the solids fraction, by separating the solids fraction odours can be reduced (Agriculture Development Guidelines Committee, 2007). Removing the solids from the storage facilities also allowed for more storage capacity of the liquid fraction. However, since regulations in Manitoba also prohibit application of manure during the winter (November 10 to April 10) the slurry solids will likely have to be stored for a period of time after centrifuging. Slurry solids have a low carbon to nitrogen ratio, high moisture content, and a high bulk density; therefore, if slurry solids were stored or spread directly onto a field the slurry solids would be prone to nitrogen losses through ammonia volatilization or nitrate leaching and would be quite odorous (Dinuccio et al., 2008; Hansen et al., 2006; Sorensen and Rubaek, 2012). Adding a bulking material and composting the slurry solids could create a higher valued marketable product that adds organic matter to soils, would make it easier to apply without odours, and distributes nutrients evenly. The process of composting converts the available ammonium into a stabilized form of organic nitrogen that provides a slow release fertilizer. Research also suggests that the process of composting can establish a diverse

population of microorganisms, and when the compost was applied the microbial population can reduce the incidence of plant diseases and parasitic nematodes (Molina et al., 2014; Thoden et al., 2011; Xu et al., 2012)

Turned windrow composting has generally been considered the preferred method of choice for producing a marketable compost product in a six month to one-year time frame that was also cost effective (Environment Canada, 2013). However, there are limitations to this composting technology and there has been limited research that looks at the suitability of composting slurry solids on-farm using turned windrow technology. Slurry solids can be a difficult feedstock material to handle for composting due to its properties including small homogeneous size, available nitrogen, and high moisture content making it prone to extensive greenhouse gas and nitrogen gaseous losses if not properly managed (Imbeah, 1997). Biosolids or sewage sludge produced from water treatment facilities have similar characteristics and Environment Canada (2013) recommends using aerated composting technology to compost these types of material due to better control managing aeration and odour, less time required to produce mature compost, and to have better success meeting the thermophilic temperature requirements described as the Process to Further Reduce Pathogens (PFRP) in the Canadian Council of Ministers of the Environment (CCME) Guidelines for Compost Quality (2005). However, aerated composting technology requires extensive expertise and there are high capital and operating costs compared to turned windrows. Composting on-farm generally has a larger buffer between neighbouring odour receptors so this may not be a major issue in this situation. Therefore, turned windrow composting of slurry solids may be a feasible option but requires investigation if thermophilic temperatures can be reached and maintained to meet CCME Guidelines for Compost Quality (2005) and if the composting process can be managed throughout the winter to produce a mature compost that can

be applied in the spring.

In addition, prior to recommending centrifugation and winter composting of separated pig slurry solids to be used as a management practice in Manitoba, information on identifying and measuring potential greenhouse and nitrogen gaseous losses throughout the composting process is required. Limiting the emission of these gases reduces the contribution of greenhouse gases from agriculture and keeps as much nitrogen as possible within the compost to be used as a slow release fertilizer. The composting process produces carbon dioxide (CO₂) and water (H₂O), notable greenhouse gases, by aerobic microbial organisms as they utilize the organic material as a food source. Greenhouse gas life-cycle analysis considers CO₂ emitted from composting a biogenic source, the amount of CO₂ emitted during the composting process should be equivalent to the amount required to produce the organic material used in the composting process, and thus limiting production or emission of CO₂ from composting is not required (Amlinger et al., 2008). Ideally, composting would only produce CO₂ and H₂O but previous research from composting has shown losses of other greenhouse gases and nitrogen can be extensive if the composting process was not managed properly (Larney et al. 2006; Amlinger et al., 2008).

The quantity of CO₂ emitted can be an indicator of the rate of decomposition. The highest CO₂ fluxes occur during the active stage, when conditions for aerobic microbial activity are met including optimal ranges of carbon to nitrogen ratio (C:N), moisture content, oxygen content, and pH. Environment Canada (2013) summarized optimal ranges for each of these parameters based on previous research and practical applications that can generally be used for most feedstock materials. The suggested optimal starting range for C:N was between 20 to 30 (Environment Canada, 2013). Above 30 too much carbon can be present and the rate of decomposition can be slowed, below 20 there can be too much nitrogen present leading to nitrogen gaseous losses and

potential odours. The optimal moisture content during the active stage of composting was suggested to be between 500 to 600 g kg⁻¹ (as-is basis); below this range there may not be enough water to provide an environment for microbial populations to survive, therefore, this can slow the rate of decomposition (Environment Canada, 2013). Above this range water can displace oxygen in the pore space and can cause the windrow to go anaerobic after the microbial organisms use up the limited oxygen. Aerobic microbial organisms require oxygen to decompose organic materials. Oxygen content should be maintained above 130 mL L⁻¹ and turning or increasing pore space may be required if below 100 mL L⁻¹ (Environment Canada, 2013). As mentioned, oxygen content has been associated with the moisture content, in situations with high moisture content, water can displace oxygen in the pore space. However, the windrow must also have the proper structure and bulk density that would allow pore space to provide oxygen and gas transfer. When initiating the compost process, the bulk density of the mixed feedstock materials should be < 475 kg m⁻³ to allow for passive aeration as well as insulation for retaining heat and moisture, this value can have a wide range and can be variable throughout the windrow (Environment Canada, 2013). The pH should not be a concern unless dealing with feedstock materials that are either acidic or basic, but the microbial population can function best and nutrients should be available when pH was within 6.5 to 8 (Environment Canada, 2013). If these conditions were met the microbial population will be active and this produces heat. During the active stage of composting, thermophilic microbial populations can be present and rapidly decompose organic material while reaching temperatures between 50 to 60 °C.

As the composting process continues CO₂ fluxes decrease as the conditions inevitably change as the feedstock material becomes compost. The C:N should decrease as the carbon decomposes and gets emitted as CO₂. The microbial activity should slow, and the moisture content

can be reduced. The oxygen content can increase as there is less of a demand for oxygen and the bulk density will increase as the volume of material becomes reduced. Temperatures also decline as the activity slows, but this allows microorganisms that prefer cooler temperatures, mesothermic and isothermic microorganisms, to establish creating a diverse population and this indicates the curing/maturation stage (Environment Canada, 2013).

Other greenhouse gas and nitrogen gases can be produced and emitted throughout the composting process. These gases can potentially be reduced through choices in management practices. Methane (CH_4) can be produced by methanogenic bacteria when anaerobic conditions are present; these conditions exist when excess moisture or rapid microbial activity deplete oxygen reserves without being adequately restored. Small pockets of methane can be produced without being emitted from the windrow because methanotrophic bacteria within the compost utilize methane for energy, however as the concentration of CH_4 builds up it can slowly be emitted or CH_4 fluxes can also peak during turning of a windrow or directly after turning as pathways for CH_4 fluxes become available (Ahn et al., 2011; Fukumoto et al., 2003; Hao et al., 2001; Sommer and Moller 2000).

Nitrous oxide (N_2O) fluxes usually occur after the active stage of composting when there have been reductions in temperature and available ammonium-nitrogen ($\text{NH}_4^+\text{-N}$) (Ahn et al., 2011; Fukumoto and Inubushi 2009; Fukumoto et al., 2012; Hao et al., 2001; Hao et al., 2004). Fukumoto and Inubushi (2009) have attributed this type of N_2O production as incomplete nitrification. Generally, feedstock material contains nitrogen in the form of $\text{NH}_4^+\text{-N}$; under optimal conditions this can undergo nitrification to nitrate/nitrite-nitrogen ($\text{NO}_3^-/\text{NO}_2^-\text{-N}$) and then assimilation to organic nitrogen. However, if ammonium-oxidizing bacteria were present to convert NH_4^+ to nitrite (NO_2^-) but nitrite-oxidizing bacteria populations were lagging then there

could be a build-up of NO_2^- that converts to N_2O and potentially nitric oxide (NO) when oxygen was not limited. Another potential production of N_2O could be through the process of complete nitrification and then denitrification (Fukumoto et al. 2003; Fukumoto, 2012). This situation can happen when nitrification occurs on the surface of the windrow where temperatures are cooler and oxygen was available, this can build up nitrate (NO_3^-) prior to assimilating to organic N. Turning the windrow can then turn the build-up of NO_3^- into the center of the windrow where oxygen was limited and N_2O can be produced as a by-product of denitrification.

Ammonia (NH_3) fluxes were found to occur during the start of the active stage of composting when NH_4^+ -N was available, it has been noted that factors such as low moisture content, increased turning or aeration, and high pH that converts NH_4^+ to NH_3 can increase the quantity and frequency of NH_3 fluxes (Elwell et al., 2002; Fukumoto et al., 2003; Fukumoto and Inubushi, 2009; Fukumoto, 2012; Hao and Chang, 2001; Jiang et al., 2011; Larney et al., 2006; Maeda et al., 2009; Paillat et al., 2005). Emission of NH_3 to the atmosphere not only can result in loss of nitrogen from the final compost product, but atmospheric NH_3 can also contribute to acid rain and acidification of the terrestrial surrounding (Krupa, 2003).

Nitrogen oxide gas emissions (NO_x) were also a concern in the atmospheric environment as they readily react to form acid rain, photochemical smog, and ozone depletion (Hao and Chang, 2001). Other research has indicated NO may be produced under multiple processes including nitrification, denitrification, chemo-denitrification, or self-decomposition of nitrous acid (Hao and Chang, 2001; Fukumoto, 2012); however, very little research was available regarding emission of NO and nitrogen dioxide (NO_2) during composting. In the one study by Hao and Chang (2001) they measured elevated NO concentrations at the beginning of the composting process. They found the production of NO to be linked with the concentration of NO_2^- when the available NH_4^+

underwent rapid nitrification under aerobic conditions. Once NO was produced it could readily be oxidized into NO₂ in the atmosphere; however, Hao and Chang (2001) did not measure an increase in NO₂ concentration on the surface of the windrow despite measuring NO.

In order to verify the relationships between gases monitored and the biological, chemical or mechanical processes throughout the composting process a system capable of nearly continuous multi-gas analysis was required. For this purpose, long-term automated chambers (LICOR BioSciences Inc., Lincoln, NB) and a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer (DX4015, Gasmeter Technologies Inc., Helsinki, FIN) system was utilized to measure emissions of CO₂, CH₄, N₂O, NH₃, NO, and NO₂. Chapter 2 identified the potential and challenges for utilizing the automated chamber and FTIR system for measuring gaseous emissions from a compost windrow. There was a need to use the system throughout the composting process to determine if the gases measured correspond with the compost properties. With this information, management practices can be used to reduce greenhouse and nitrogen gaseous losses while composting slurry solids. To verify the ability of the automated chamber and FTIR system to capture gaseous emissions the total C and N emissions can be compared to total C and total N losses calculated by mass balance.

The overall quality of the final compost product can be dependent on both the feedstock material that goes into it, as well as the management throughout the composting process. In order for the compost to be beneficial for soil when applied, compost must meet maturity requirements and be within a range of various agronomic parameters. Maturity was required as it was an inherent characteristic of compost quality; the definition of composting requires that the end result is a mature product called compost (CCME, 2005). If immature compost was applied to soil it could potentially contain phytotoxic compounds that could adversely affect plant growth such as high

$\text{NH}_4^+\text{-N}$, non-immobilized trace elements, or organic acids (Paul and Geesing, 2009). Additionally, if immature compost with a high C:N was applied to the soil, soil microorganisms can use N in the soil to further decompose the carbon, reducing available N for plants to use (Larney et al., 2006; Paul and Geesing, 2009). To determine if the final product was mature, the potential for $\text{NH}_4^+\text{-N}$ phytotoxicity can be assessed by the ratio of $\text{NH}_4^+\text{-N}$ to $\text{NO}_3^-/\text{NO}_2^-\text{-N}$; the degree of decomposition can be assessed using the temperature of the windrow and the C:N. Agronomic parameters that are usually assessed to determine compost quality include nutrient content including total nitrogen, total phosphorus, and total potassium (K), organic matter, chemical properties (pH, electrical conductivity (EC), total sodium (Na)), and ease of use (bulk density (BD), moisture content (MC)). Each of these parameters have a range in which compost should be within; however, the actual performance of compost in the soil was also dependent on soil type, application rate and soil management (Carter et al., 2004; Paul and Geesing, 2009; Rynk, 1992). A producer must know the overall quality of compost in order to use or market the product. Therefore, the objectives of this study were to:

- 1) Determine if composting throughout the winter can be a suitable on-farm management option for separated pig slurry solids by:
 - a. Determining if thermophilic temperatures can be reached to meet PFRP criteria and if the composting process can be maintained throughout the winter to produce mature compost ready for application in the spring.
 - b. Assessing the overall quality and consistency of composted pig slurry solids in terms of compost maturity indicators (temperature, C:N and $\text{NH}_4^+\text{-N}:\text{NO}_3^-/\text{NO}_2^-\text{-N}$) and agronomic parameters (Total N, Total P, Total K, Organic Matter, pH, EC, Total Na, BD, and MC).

- 2) Assess the capability of the automated chamber and FTIR system to be used as a monitoring tool for determining the quantity and timing of GHG and N gaseous losses from composting pig slurry solids with respect to corresponding physical properties and management of the windrow, as well as compare total C and N emissions to total C and N calculated by mass balance.

3.3 Materials and Methods

3.3.1 Feedstock Sources

The pig slurry was from a conventional farrow to finish pig barn, near St. Adolphe, Manitoba, with a pull-pit collection system that drained into a wet well. The slurry solids were obtained by centrifugation of pig slurry without addition of a polymer using a two-phase clarifying decanter (ALSYS 20, Alfa Laval Inc., Scarborough, ON). The slurry solids were stockpiled over a period of two to four weeks before use in one of the three trials in this experiment. A longer period of stock piling was not possible because the temperatures of the piles rose above 50 °C and there was concern for considerable NH₃ volatilization.

Wheat straw was added to the slurry solids as a carbon source and bulking material to facilitate composting. Wheat straw was a readily accessible carbon source; the wheat straw bales were from the University of Manitoba's Glenlea Research Station near Glenlea, Manitoba. The bales had been stored uncovered for approximately one year; because the bales were aged and had absorbed moisture the bales were ideal for use in composting because they had lost value as a bedding material.

3.3.2 Compost Setup and Windrow Management

Three compost trials were setup in fall 2011: Trial 1 on September 14, 2011 or day of year (DOY) 256, Trial 2 on October 27, 2011 (DOY 299), and Trial 3 on November 15, 2011 (DOY 318). Each trial had one windrow comprised of a mixture of slurry solids and wheat straw. Due to the previously mentioned inability to stockpile the slurry solids for a long period of time prior to composting, material was only available to setup up one windrow per trial. The windrows were setup in a field east of the pig barn from which the raw slurry came from. There was no shelter from the wind from any other direction. The windrows were setup in a north-south direction. The base of the composting site was topsoil, Humic Gleysol, of the poorly drained Osborne soil series of the Red River Association (Land Resource Unit, 1999).

Grab samples of the feedstock materials, straw and slurry solids, were taken as each material was added to the feedmixer or windrow, approximately thirty samples of each material 1 L in size were taken. The grab samples were mixed and quartered down to a 2 L subsample. Re-mixing the grab samples and repeating the quartering process obtained a duplicate sample. The two samples were frozen at -18 °C prior to shipping to A&L Laboratories (London, ON) and analyzed for BD, moisture content (MC), C:N, Total nitrogen (N), Total phosphorus (P), and Total potassium (K). There was low variability among all average values from the analysis of feedstock materials, which is ideal (Table 3.1). The Total N values for the slurry solids used in the Trials were similar, despite being produced at different times, this indicated that N losses while the slurry solids were stockpiled were consistent.

Table 3.1 Characteristics of feedstock materials, straw and separated slurry solids.

Material	Trial	BD	C:N	MC	Total N	Total P	Total K
		kg m ⁻³	Ratio	g kg ⁻¹			
Straw	1	43(42, 43)	37(39,34)	139(139,138)	13.4(12.2,14.5)	6.7(10.7,2.6)	13.4(13.3,13.4)
	2	32(33,30)	93(98,87)	127(131,123)	5.6(5.3,5.9)	1.0(0.9,1.0)	20.6(20.9,20.2)
	3	33(33,33)	51(49,53)	193(166,219)	10.0(10.4,9.6)	1.2(1.1,1.2)	14.1(15.2,12.9)
	Avg.	36 (2)	60 (11)	153 (15)	9.6 (1.5)	2.9 (1.6)	16.0 (1.5)
Slurry Solids	1	325(312,337)	15(15,15)	591(591,591)	29.3(29.4,29.1)	14.3(26.3,2.2)	9.2(5.4,12.9)
	2	369(376,362)	16(16,15)	650(646,653)	27.9(25.7,30.1)	20.1(19.3,20.9)	5.4(5.6,5.1)
	3	356(360,352)	11(11,11)	524(488,559)	30.8(29.9,31.6)	23.5(22.8,24.1)	6.9(7.1,6.6)
	Avg.	350 (9)	14 (1)	588 (25)	29.3 (0.8)	19.3 (3.6)	7.1 (1.2)

Values for Trials 1, 2, and 3 are the mean of the two sub-sample values shown in parenthesis, the average (Avg.) being the mean of six samples ± 1 standard error in parenthesis. Values for bulk density (BD) and moisture content (MC) are reported on an as-is basis and those for total nitrogen (N), phosphorus (P), and potassium (K) on a dry mass basis.

Trials 1 and 2 had the slurry solids, straw, and water weighed and mixed in a feedmixer (900T, Supreme Int., Wetaskiwin, AB) and then setup into windrows using the side conveyor for unloading. Trial 3 was setup by placing the slurry solids on top of a windrow of straw and mixing them using a front mounted paddle auger windrow compost turner (Brown Bear PA35D Compost Aerator, Brown Bear Corp., Corning, IA). In total, Trial 1 had 3,120 kg of straw and 2,590 kg of slurry solids; Trial 2 had 2,550 kg of straw and 2,383 kg of slurry solids, on a dry mass basis. Feedstock materials were not weighed in Trial 3; the estimated ratio of feedstock materials was calculated using the C:N, bulk density (BD), and volume of the starting windrow. The estimated starting amount of feedstock materials in Trial 3 was 1,590 kg of straw and 1,572 kg slurry solids, on a dry mass basis. The mixing ratios of straw to slurry solids, on a dry mass basis, for Trials 1, 2, and 3 were 1.2, 1.1, and 1.0, respectively (Table 3.2). Trial 2 had the highest starting MC, which raised the BD (Table 3.2). The starting C:N for all three Trials were very similar and in the ideal starting range (Table 3.2) The starting windrow dimensions of Trials 1, 2, and 3 had a length of 40.1 m, 37.5 m, and 30 m, respectively; Trial 1 and 2 had a width of 2.37 m, Trial 3 had a width of 3 m; and all windrows had a height of 1 m.

Table 3.2 Added amount of feedstock materials, straw and slurry solids, and characteristics of starting mixture at the start of Trials 1, 2, and 3.

Trial	Start date	Added Feedstock Material		Starting Mixture		
		Straw	Slurry Solids	BD kg m ⁻³	MC g kg ⁻¹	C:N Ratio
1	September 14, 2011	546	454	173 (8)	583 (20)	25 (0)
2	October 27, 2011	517	483	189 (6)	713 (10)	24 (2)
3	November 15, 2011	503	497	98 (5)	464 (70)	25 (2)

Values are the mean of three samples ± 1 standard error in parenthesis. Values for added feedstock material is on a dry mass basis and those for Bulk Density (BD) and Moisture Content (MC) are reported on an as-is basis.

The windrows were turned with the Brown Bear windrow compost turner on September 21 (DOY 263), October 6 (DOY 278), October 27 (DOY 299), November 3 (DOY 306), November 15 (DOY 318), November 30 (DOY 333), December 16 (DOY 349), January 10 (DOY 374), and April 11 (DOY 466) for all trials. Therefore, there were nine, six and four turnings of windrows in Trials 1, 2 and 3, respectively. After Trial 1 had been initiated the weather was warm, dry, and windy resulting in the moisture content going below 400 g kg⁻¹. Water was added in Trial 1 using a garden trickle hose on October 5 (DOY 277) and October 11 (DOY 283). When water was added in this way it appeared that some of the slurry solids “washed off”, due to the difference in particle size and the waxy texture of straw causing poor contact between the straw and slurry solids, the slurry solids leached into the soil under the windrow. Throughout the experiment, September 14, 2011 to April 11, 2012, the area received 119 mm of total precipitation (weather station located at the Trace Gas Manitoba Research Site, Glenlea MB, approximately 15 kilometers south-west of the composting site), which was less than the 30-year average (1981-2010) Climate Normal of 218 mm, for the same September to April months, measured at the Winnipeg Richardson International Airport, Winnipeg Manitoba (Environment Canada, 2014). Additional water was not added to Trial 2 or 3 windrows.

The Brown Bear compost windrow turner unit was mounted to the front of a low-speed tractor using a 3-point hitch (Figure 3.1). The blade of the Brown Bear sits slightly above the ground and pushes material into the paddle auger, which agitates and aerates the material and moves it to the side creating a windrow. The operation of this unit and the location of the site in an open field led to accidental addition of clay soil to the windrows; this occurred most notably on: November 15 (DOY 318) the Trial 2 windrow was moved using a front end loader; on November 30 (DOY 333) when snow pack caused an uneven surface that caused the turner to

scrap into the soil and turn soil into all three of the windrows; and on January 10 (DOY 374) when wind eroded soil from the field and deposited on all three of the windrows.



Figure 3.1 Trial 1 windrow, prior to turning on September 21, 2011 (DOY 263), with automated chambers set up, as well as Brown Bear windrow compost turner in background.

3.3.3 Compost Sampling and Analysis

Throughout the experiment in-process samples were obtained immediately after a windrow was turned. Twenty random grab samples, of 1 L each, were taken across the windrow approximately 10-60 cm below the surface. The grab samples were then mixed and quartered down to obtain three 2 L sub-samples and three 200 g sub-samples. In this experiment the remainder of the material was not returned to the respective windrow, there was also additional material taken, total amount undocumented, with the intent to do additional analysis, but was not conducted. All samples were frozen at $-18\text{ }^{\circ}\text{C}$ prior to analysis. The three 2 L sub-samples were shipped to A&L Laboratories (London, Ontario) for analysis of BD, C:N, total organic matter, total organic carbon, and total N, P, K, and Na. The three 200 g sub-samples were cut using scissors until almost all of the straw was 1 cm or less in length and analyzed for MC, EC, pH, extractable $\text{NH}_4^+\text{-N}$ and NO_3^-

/NO₂⁻-N at the Soil Ecology Laboratory (Soil Science Department, University of Manitoba, Manitoba). For Trials 2 and 3, only the initial samples and the final samples were sent to A&L Laboratories for analysis.

Bulk density was measured as-received in kg m⁻³; a 20 cm burette was filled with as-received sample, then a rubber ball was placed on top and the burette was dropped from 15 cm to reproduce settling in a compost windrow, the volume and mass was recorded (Test Methods for the Evaluation of Composting and Compost (TMECC) 3.10-D; Thompson et al., 2001). The C:N ratio was determined from Total N divided by total organic carbon (TMECC 5.02-A; Thompson et al., 2001). Total N, in g kg⁻¹ on a dry mass basis, was determined using the Dumas method (dry combustion oxidation) of dry-milled samples (TMECC 4.02-D; Thompson et al., 2001). Total organic carbon, in g kg⁻¹ on a dry mass basis, was calculated as a percentage of total organic matter determined by loss on ignition of dry-milled material ignited at 550 °C (TMECC 3.02-B; Thompson et al., 2001). Total P, K, and Na, in g kg⁻¹ on a dry mass basis, were determined from air-dried milled samples digested with nitric acid (TMECC 4.12-B nitric acid digestion; Thompson et al., 2001) then inductively coupled plasma (ICP) spectrometry (TMECC 4.14; Thompson et al., 2001) using TMECC 4.03-A, 4.04-A, and 4.05-Na protocols (Thompson et al., 2001), respectively. Moisture content in g kg⁻¹ was determined by mass loss of a 10-15 g sample dried at 70 °C (TMECC 3.09-A; Thompson et al., 2001). Electrical conductivity (TMECC 4.10) and pH (TMECC 4.11) were determined using a 50 g as-received sample mixed with 250 mL of reverse osmosis water using a reciprocating shaker at 150 excursions per minute for 30 minutes (Thompson et al., 2001). Electrical conductivity was measured in mS cm⁻¹ using a CDM2e conductivity meter (Radiometer Medical ApS, Bronshoj, Denmark) with CDC3x4 Conductivity Cell (Radiometer Analytical S.A., Villeurbanne, France); pH was measured using a 720A

benchtop meter (Orion Research Inc., Beverly, Massachusetts) with a Thermo Scientific electrode (Chelmsford, Massachusetts). Extractable NH_4^+ -N and $\text{NO}_3^-/\text{NO}_2^-$ -N were determined from 30 g as-received sample shaken with 150 ml 2M KCl for one hour using a reciprocating shaker (150 e.p.m.; TMECC 4.02-B and 4.02-C; Thompson et al., 2001). The concentration of NH_4^+ -N and $\text{NO}_3^-/\text{NO}_2^-$ -N in extracts was determined using the Berthelot reaction (alkaline phenol and hypochlorite solution) and cadmium-reduction azo-dye method using colorimetry and an Autoanalyzer II (SEAL Analytical Ltd., Hampshire, United Kingdom). Results are reported on a dry mass basis sample as mg N kg^{-1} .

For each of the sampling dates the three samples were treated as replicates, the average and standard error were determined for each parameter. The final compost samples were taken on April 11, 2012 (DOY 466). Statistical analyses were performed on the results of the final compost samples with SigmaPlot Systat Software (SigmaPlot for Windows Version 11.0, Germany). A one-way analysis of variance (ANOVA) was used to compare the final parameters for each of the Trials. A Tukey test, for pairwise multiple comparison of means, was used to determine significant differences ($P < 0.05$). Data was tested for normality and equal variance (Shapiro-Wilks). Data for C:N, pH, and total Na failed the normality test, therefore a Kruskal-Wallis one-way analysis of variance on ranks was used.

3.3.4 Compost Temperature, Air Temperature, and Oxygen Concentration

The temperature of the compost windrow was monitored every half-hour throughout each trial using thermistors or thermocouples attached to a CR1000 data logger and multiplexer (Campbell Scientific Canada, Edmonton, AB). Initially temperature was measured using the average of at least four 109-Thermistors (Campbell Scientific Canada, Edmonton, AB) placed in the core of

Trial 1 and 2 windrows. These thermistors were found to give inaccurate readings due to poor signal cable connections caused by breakage of the wiring from the heat produced while composting; they were also difficult to consistently place them at the same location and depth throughout the windrow. On December 16 (DOY 349), the thermistors were replaced with in-house made Type T thermocouples that were inserted into a wooden dowel; four Type T thermocouples along the dowel at 20 cm spacing. Four dowels were placed in a windrow, the tip of the dowel started at the core and angled out perpendicular to the side of the windrow. The temperature of the windrow was determined by averaging temperatures of the four dowels at the four depths. The average temperature of the windrow in Trial 3 was monitored approximately every week using three 4 ft compost thermometers (Reotemp Instrument Corp., San Diego, California) inserted into the core of the windrow.

Air temperature was measured using a 109-Thermistor until it was replaced by an in-house made Type T-thermocouple on December 16 (DOY 349). The sensor was placed at a height of 1.5 meters and was sheltered from sunlight and wind. The 30-year average (1981-2010) Climate Normal air temperature was measured at the Winnipeg Richardson International Airport, Winnipeg Manitoba (Environment Canada, 2014).

Measurements of oxygen concentration were taken using an OT-21 oxygen monitor (Geneq Inc. Montreal, Quebec). Prior to turning, each windrow in Trials 1, 2, and 3 had oxygen concentrations measured at three locations in the core of the windrow.

3.3.5 Gas Flux Measurement

As described in Chapter 2, automated chambers (Long-term Automated Chambers, LICOR BioSciences Inc., Lincoln, NB) and a FTIR multi-gas analyzer (DX4015, Gasmeter Technologies

Inc., Helsinki, FIN) were used to determine gas fluxes from the windrows. The two components were placed in series to create a closed loop system that collected and measured the accumulation of gas emitted from the windrow when the chamber lids were deployed. The chambers were set up on the top of a windrow and were inserted 8 cm into the surface. The FTIR was capable of simultaneously measuring CO₂, CH₄, N₂O, NO₂, NO, and NH₃ volumetric concentrations.

Eight chambers were initially set up on the Trial 1 windrow on September 19 (DOY 261); on October 3 (DOY 275) one chamber stopped opening and closing due to corrosion of the metal pins that connected the signal cable of the automated chamber with the multiplexer that signalled the opening and closing of the chamber; the connection was not properly tightened allowing water and NH₃ to corrode the metal. When Trial 2 was initiated on October 27 (DOY 299) four chambers were moved from Trial 1 to the Trial 2 windrow. On November 10 (DOY 313), one of the chambers on Trial 2 stopped opening and closing, again due to corrosion. Connecting the signal cable correctly to ensure a tight seal was crucial to prevent corrosion. Gas flux data was not measured for Trial 3.

A gas flux measurement could only be taken from one chamber at a time. The sequence was set up to measure gas fluxes from all the Trial 1 chambers in series then switched to measure from all Trial 2 chambers. The time the chamber was closed, referred to as deployment time, and the interval between measurements can be adjusted based on the rate of gas flux and purge time required. On September 19 (DOY 261), chamber deployment time was 3 minutes and purge time was 12 minutes. On October 3 (DOY 275), purge time increased to 17 minutes. On March 15 (DOY 439), purge time increased to 27 minutes allowing a total of 30 minutes between measurements. Increasing the purge time allowed more time for the gases that had accumulated in the system to go back down to background levels. As noted in Chapter 2, there was not sufficient

purge time for NH_3 to purge from the system at the start of this experiment when NH_3 was emitted; possibly causing error in gas flux estimations. It was not necessary to continue increasing the purge time late in the experiment; however, increasing the total time between gas flux measurements to 30 minutes aligned gas flux measurements with temperature and wind measurements so that correlation analysis could be done between gas fluxes and these parameters; see Chapter 5 for wind data.

For Trial 1, gas fluxes were monitored from September 19 (DOY 261) to April 13 (DOY 468), a total of 207 days. Trial 2 gas fluxes were monitored from October 28 (DOY 300) to April 13 (DOY 468), a total of 168 days.

3.3.6 Gas Flux and Cumulative Emission Estimation

The procedure for calculating gas fluxes was given in detail in Chapter 2. In brief, the Ideal Gas Law ($PV=nRT$) was used to convert gas concentration (ppm) in the total volume of the automated chamber and FTIR system into mass of carbon (g-C) or nitrogen (g-N). Then gas fluxes were estimated in g-C or $\text{N m}^{-2}\text{s}^{-1}$ from the linear change in gas concentration over time and the surface area of the chamber collar.

Each chamber was a pseudo-replicate unit; daily gas flux was calculated by averaging all fluxes from a chamber over the course of a day. The average and one standard error were determined for each day from the daily gas flux for individual chambers.

Adding the daily gas flux for each chamber and then averaging those values calculated cumulative emissions for each windrow in g-C or N m^{-2} . When a chamber was taken out of the Trial the cumulative sum was averaged at that point and that value was used to continue for the next interval. Equipment maintenance or delay in setting up the equipment after turning resulted

in a few days when monitoring did not occur. Linear interpolation of gaps in the daily gas flux data was used for the following periods: September 23-27 (DOY 265-269), September 29 (DOY 271), November 7-9 (DOY 310-311), January 18-19 (DOY 382-383), January 22-February 2 (DOY 386-397), and March 19 (DOY 443). Linear interpolation was not used from September 19-23 (DOY 261-265), October 27 (DOY 299), and March 28-April 11 (DOY 452-466), because a turning event occurred in between that time, differences in gas fluxes may have been extensive. Linear interpolation was not used from February 4 to March 15 (DOY 399-439). During this period gas fluxes were not monitored because air temperature was very cold (< -30 °C) preventing operation of the FTIR and snow/ice interfered with chamber opening and closing on a frequent basis. It was assumed that there were zero gas fluxes during this time based on the near 0 °C compost temperature of Trial 1 and 2 and the relatively low gas fluxes prior to and after this time period.

The mass of gases emitted (g-C or g-N) was calculated by multiplying the surface area of the windrow by the daily average gas flux and then adding these values for each day during the experiment duration. The total surface area of the windrow was calculated after each time the windrow was turned; using the formula for surface area of a windrow with a shape in between an oval and a trapezoid, as described in Paul and Geesing (2009). The total mass of gases emitted was also expressed based on the initial dry mass of the windrow (kg C or N Mg^{-1}). The as-is starting mass of the windrow was known because it was weighed in the feedmixer, analysis of moisture content allowed estimation on a dry mass basis. The sum of g-C from CO_2 and CH_4 emissions were used to calculate total g-C of gases emitted, and then the percentage of each contributing to the total was determined. Similarly, the sum of g-N from N_2O , NO , and NH_3 emissions were used

to calculate total g-N of gases emitted; then the percentage of each contributing to the total was determined.

Statistical analysis was not conducted to compare the gas flux emissions from Trial 1 and Trial 2. The design of the experiment allowed the number of chambers used to measure gas fluxes to change over the course of the experiment, thus the chambers could not be treated as pseudo replicates. The purpose of this experiment was not intended to compare gas emissions from the Trials but to observe the gas fluxes in response to the composting properties.

3.3.7 Estimated Losses of Mass, Carbon and Nitrogen Based on Mass Balance

Carbon and nitrogen loss based on mass balance was estimated by subtracting the total C or total N within the final material from the total C or N within the starting material. Total C was calculated by multiplying the initial or final windrow mass, on a dry mass basis, by the initial or final value of organic carbon determined in the samples sent for analysis. Total N was calculated similarly by multiplying the initial or final windrow mass, on a dry mass basis, by the initial or final value of Total N determined in the samples sent for analysis. The final mass was calculated based on estimations of volume, calculated using the formula for surface area of a windrow with a shape in between an oval and a trapezoid (Paul and Geesing, 2009), and bulk density and moisture content analysis. The values for mass, total C and total N were expressed as the amount per initial windrow dry mass (g Mg^{-1}). The loss of mass, loss of total C and loss of total N was also expressed as a percentage of the initial dry mass, initial total C, or initial total N, respectively. The total mass of gaseous emissions measured were also expressed as a percentage of initial total C and initial total N, as well as the percentage of total C or N loss.

3.4 Results

3.4.1 Air Temperature and Precipitation During the Trials

From September 2011 through to April 2012 the average monthly air temperature was slightly above the 30-year (1981-2010) Canadian Climate Normals for the Winnipeg Richardson International Airport, located approximately 35 kilometers north of the composting site (Environment Canada, 2014). Average daily air temperature was around 14 °C throughout September into October with daytime highs reaching above 25 °C (Figure 3.2). Starting midway through November, average daily temperatures dropped to 0 °C and steadily declined to -10 °C in January, with a few daytime lows reaching -30 °C. Average daily temperature started to increase to 6 °C after mid March into April.

Precipitation during these months was comparatively lower than the 30-year (1981-2010) Canadian Climate Normals for the Winnipeg Richardson International Airport (Environment Canada, 2014), especially during the month of October (Figure 3.2). Zero precipitation was measured for the months of December 2011 and January 2012; these values may be underestimated due to equipment failure or rounding errors, but from observation there was very little snow during this period.

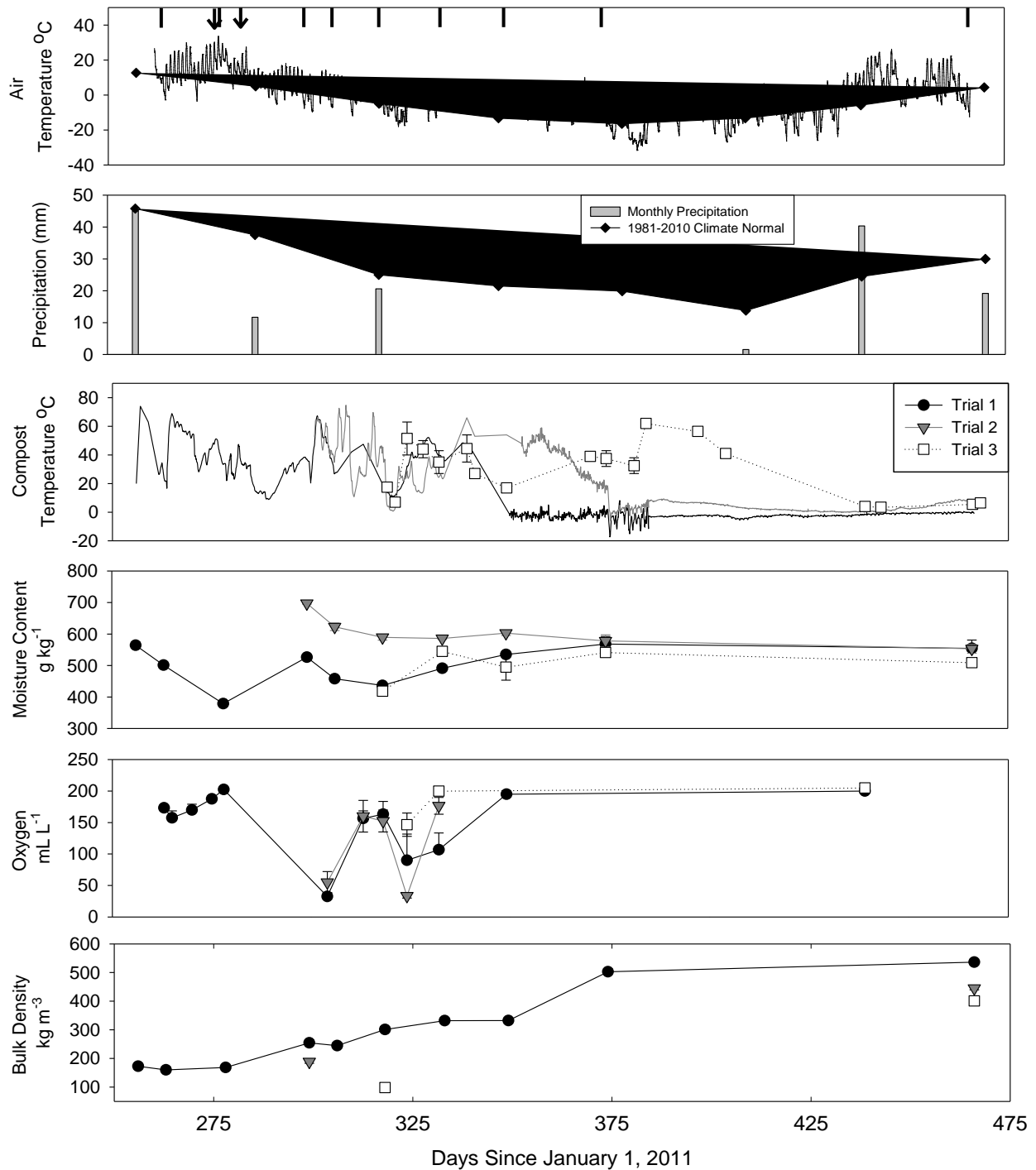


Figure 3.2 Measured air temperature and monthly precipitation and 30-year (1981-2010) Winnipeg Richardson International Airport Climate Normals for average air temperature and precipitation, compost windrow temperature, moisture content (MC), oxygen concentration, and bulk density (BD) during the Trials 1, 2, and 3 from September 14, 2011 (DOY 257) to April 13, 2012 (DOY 468). Values for air and compost temperature are the mean of all measurements recorded and the values for MC, oxygen, and BD are the mean of three samples and bars where shown are ± 1 standard error. MC and BD are reported on an as-is basis. Bars and arrows indicate windrow turning and water added, respectively.

3.4.2 Compost Properties During the Trials

3.4.2.1 Compost Temperature

The thermophilic period for Trial 1, indicated by the span of time in which average temperature was above 50 °C, lasted 72 days, from September 15 (DOY 257) to November 26 (DOY 329) (Figure 3.2). Using only the highest recorded temperature to determine if PFRP requirements were met, the temperature was above 55 °C for 42 non-consecutive days, in which the windrow was turned 5 times. At the start of that thermophilic period, shortly after initiating the windrow, temperatures slowly declined, adding water on October 5 (DOY 277) and turning the windrow on October 6 (DOY 278) did not increase the compost temperature, temperatures stayed below 20 °C. After adding more water on October 11 (DOY 283) and turning the windrow on October 27 (DOY 299) the windrow temperature rapidly increased to above 50 °C. After November 26 (DOY 329), temperatures were in the mesothermic stage around 40 °C; however, these temperatures immediately dropped to and stayed at 0 °C when the windrow was turned on December 16 (DOY 349) and air temperatures were -10 °C. In the spring, after the windrow thawed, it was turned on April 11, 2012 (DOY 466); the temperature did not increase above ambient temperatures after turning.

The compost temperature for Trial 2 indicated the thermophilic period was 59 days from October 28 (DOY 300) to December 26 (DOY 359) (Figure 3.2). During this time the highest temperature measured was above 55 °C for 37 non-consecutive days during which it was only turned four times; therefore, did not meet CCME Guidelines for Compost Quality PFRP criteria. During the thermophilic period, the compost temperature decreased to below 40 °C after turning the windrow on November 15 (DOY 318), but then rapidly increased back up to above 50 °C after the windrow was turned again on November 30 (DOY 333). Compost temperature gradually

declined into the mesothermic stage after December 26 (DOY 359), but stayed above 17 °C until the windrow was turned on January 10 (DOY 374) when air temperature was 3 °C, at which point the compost temperature then dropped to 0 °C. The temperature slowly recovered and increased to 10 °C on January 23 (DOY 387), but declined thereafter and only reached ambient air temperature as temperatures increased in the spring and the windrow was turned.

Compost temperature for Trial 3 was measured using thermometers, average temperature above 50 °C was measured on November 21 (DOY 324), January 20 (DOY 384), and February 2 (DOY 397), therefore it was assumed the thermophilic period lasted 73 days between November 21 (DOY 324) and February 2 (DOY 397) (Figure 3.2). Without frequent monitoring it was difficult to determine exactly how many days were above 55 °C, however, it was only turned three times during the assumed thermophilic period, therefore CCME Guidelines for Compost Quality PFRP criteria were not met. Compost temperature gradually declined from 41 °C, on February 9 (DOY 404), to 4 °C, on March 14, 2012 (DOY 439) and did not re-heat above ambient air temperature after it was turned in the spring.

3.4.2.2 Compost Moisture Content

Moisture content for Trial 1 started within the optimal range at 564 g kg⁻¹ and then rapidly declined to 379 g kg⁻¹ by October 6 (DOY 278) (Figure 3.2). Reduced moisture content corresponded with reduced temperature of Trial 1. After water was added on October 5 (DOY 277) and October 11 (DOY 283) moisture content returned to 526 g kg⁻¹ by October 27 (DOY 299) and was maintained between 500 to 600 g kg⁻¹.

Moisture content for Trial 2 started above the recommend range, at 700 g kg⁻¹ (as-is basis), because too much water was added initially. By November 15 (DOY 318) moisture content had

gradually declined to 590 g kg^{-1} and was maintained within the optimal range thereafter (Figure 3.2).

Moisture content for Trial 3 started below the recommended range, at 419 g kg^{-1} , because no water was added to the initial mixture (Figure 3.2). However, moisture content increased up to 545 g kg^{-1} by November 30 (DOY 333) with the incorporation of snow and was maintained within the optimal range thereafter.

3.4.2.3 Compost Oxygen Concentration

In Trial 1, oxygen concentration was maintained above 130 mL L^{-1} until October 6 (DOY 278) then began to decrease to 30 mL L^{-1} by November 1 (DOY 304) (Figure 3.2). Turning the windrow on November 3 (DOY 306) immediately increased oxygen concentration. On November 21 (DOY 324) oxygen concentration went down to 90 mL L^{-1} , but recovered after turning the windrow on November 30 (DOY 333) and remained around 200 mL L^{-1} thereafter.

Trial 2, started with very low oxygen concentration at 55 mL L^{-1} on November 1 (DOY 304) (Figure 3.2). After turning the windrow, oxygen content was above 130 mL L^{-1} from November 10 to 15 (DOY 313-318), but then dropped to 33 mL L^{-1} on November 21 (DOY 324). By November 29 (DOY 332) oxygen concentration had recovered to 180 mL L^{-1} . On December 16 (DOY 349) the outer layer of the Trial 2 windrow was frozen and the oxygen probe could not be inserted, therefore measurements could not be taken after that date.

Trial 3 maintained oxygen concentration above 130 mL L^{-1} throughout the trial (Figure 3.2). Data was not available on December 16 (DOY 349) because measurements were not taken.

3.4.2.4 Compost Bulk Density

Bulk density (BD) for Trial 1 started at 174 kg m^{-3} (as-is basis) and ended at 536 kg m^{-3} , Trial 2 started at 189 kg m^{-3} and ended at 445 kg m^{-3} , and Trial 3 started at 98 kg m^{-3} and ended at 401 kg m^{-3} (Figure 3.2). Trial 1 BD steadily increased until January 10 (DOY 374), 118 days after the compost process was initiated, and then remained at a steady state. Bulk density measurements were not taken throughout the composting process for Trial 1 and 2.

3.4.2.5 Compost pH

The pH of Trial 1 started at 6.8, then increased to 7.7 after the first turning event on September 21 (DOY 263), and then decreased to 7.0 after the next turning event on October 6 (DOY 278) before increasing and staying stable around 7.7 after the third turn on October 27 (DOY 299) (Figure 3.3). The pH for Trial 2 started slightly higher than the recommended range at 8.26, (Figure 3.3). After turning the windrow, the pH came down to 8.0. The pH for Trial 3 started within optimal range at 7.5 and finished within optimal range at 7.42 (Figure 3.3).

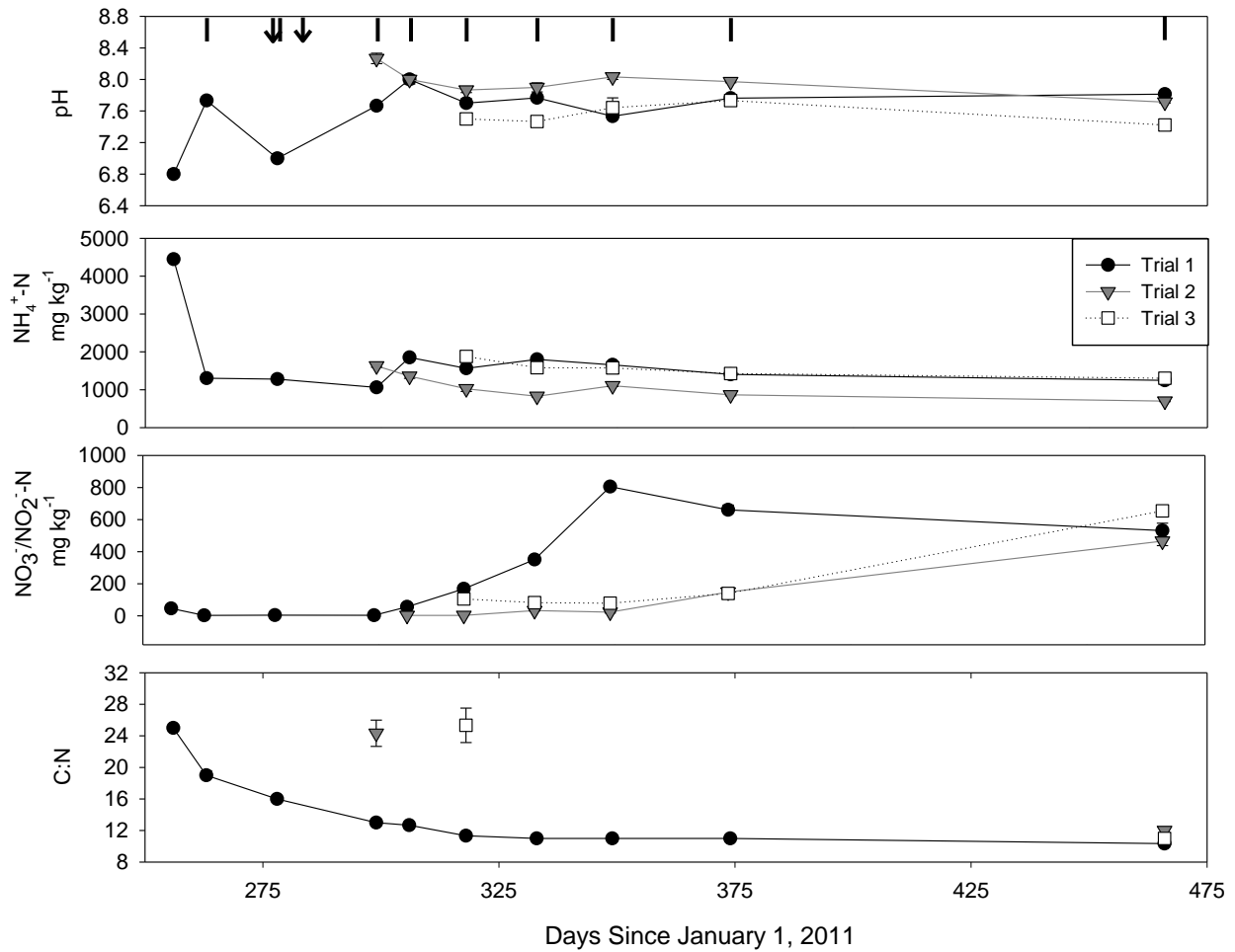


Figure 3.3 pH, ammonium ($\text{NH}_4^+\text{-N}$), nitrate/nitrite ($\text{NO}_3^-/\text{NO}_2^-\text{-N}$), and carbon to nitrogen ratio (C:N) during Trials 1, 2, and 3 from September 14, 2011 (DOY 257) to April 13, 2012 (DOY 468). Values for pH, $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-/\text{NO}_2^-\text{-N}$, and C:N are the mean of three samples and bars where shown are ± 1 standard error. $\text{NH}_4^+\text{-N}$, and $\text{NO}_3^-/\text{NO}_2^-\text{-N}$ are reported on an as-is basis. Bars and arrows indicate windrow turning and water added, respectively.

3.4.2.6 Compost Ammonium and Nitrate/Nitrite

In Trial 1, $\text{NH}_4^+\text{-N}$ started relatively high, at $4,447 \text{ mg N kg}^{-1}$ (as-is basis), compared to starting values of Trial 2 and 3, which were $1,630 \text{ mg N kg}^{-1}$ and $1,880 \text{ mg N kg}^{-1}$, respectively (Figure 3.3). However, on September 21 (DOY 263), 3 days after initiating the Trial, $\text{NH}_4^+\text{-N}$ rapidly dropped to $1,303 \text{ mg N kg}^{-1}$ and then gradually declined to $1,250 \text{ mg N kg}^{-1}$. Trial 2 and 3 $\text{NH}_4^+\text{-N}$ gradually declined to 700 mg N kg^{-1} and $1,303 \text{ mg N kg}^{-1}$, respectively.

Trial 1, 2, and 3 $\text{NO}_3^-/\text{NO}_2^-$ -N started at 46, 3, and 105 mg N kg^{-1} (as-is basis), respectively (Figure 3.3). Trial 1 $\text{NO}_3^-/\text{NO}_2^-$ -N rapidly increased from November 3 (DOY 306) until December 16 (DOY 349) where it peaked at 805 mg N kg^{-1} , 93 days after initiating the trial, and then declined slightly thereafter. Trial 2 and 3 $\text{NO}_3^-/\text{NO}_2^-$ -N started to increase after November 30 (DOY 333) and January 10 (DOY 374), 34 and 56 days after initiating the trials, respectively. Final $\text{NO}_3^-/\text{NO}_2^-$ -N for Trial 1, 2, and 3 was 531, 465, and 654 mg N kg^{-1} , respectively.

3.4.2.7 Compost Carbon to Nitrogen Ratio

The starting C:N for all the trials were within the optimal range with Trials 1, 2 and 3 at 25, 24.3, and 25.3, respectively (Figure 3.3). The C:N was measured throughout Trial 1, there was a rapid decline until November 15, 2011 (DOY 318), 62 days after initiating the windrow, when the C:N levelled off and was maintained around 12. The C:N was not measured throughout the composting process for Trial 2 and 3. The final C:N of Trials 1, 2, and 3 was 10, 12, and 11, respectively.

3.4.3 Greenhouse Gas and Nitrogen Gaseous Losses

3.4.3.1 Trial 1 Daily Mean Gas Fluxes

In Trial 1, CO_2 fluxes started low compared to later in the process after more water was added and the windrow was turned (Figure 3.4). A large CO_2 flux peaked on October 28 (DOY 300) at 1,212 g C $\text{m}^{-2} \text{d}^{-1}$. Fluxes of CO_2 declined shortly afterwards and did not rise until after the windrow was turned on November 15 (DOY 318), fluxes peaked at 487 g C $\text{m}^{-2} \text{d}^{-1}$ on November 19 (DOY 322) and another peak shortly after on November 26 (DOY 329) at 309 g C $\text{m}^{-2} \text{d}^{-1}$. After December 12 (DOY 346) fluxes were less than 10 g C $\text{m}^{-2} \text{d}^{-1}$. Therefore, the active stage of composting, considered as the span of time CO_2 fluxes were high and indicative of high microbial activity, lasted 46 days and ranged from -9 to 1,212 g C $\text{m}^{-2} \text{d}^{-1}$ with an average of 114 g C $\text{m}^{-2} \text{d}^{-1}$.

Methane fluxes occurred during the active stage and followed the trend of CO₂ fluxes. CH₄ fluxes increased immediately after turning on October 28 (DOY 300) and peaked at 1.85 g C m⁻² d⁻¹ on November 1 (DOY 304) (Figure 3.4). After turning the windrow on November 3 (DOY 306), CH₄ fluxes were minimal until November 12 (DOY 315) when there was a slight increase of CH₄ fluxes that peaked at 1.11 g C m⁻² d⁻¹ but declined prior to turning on November 15 (DOY 318). The third CH₄ flux peak increased on November 24 (DOY 327) and peaked on November 27 (DOY 330), at 1.85 g C m⁻² d⁻¹, then immediately declined after turning on November 30 (DOY 333) and remained negligible for the remainder of the Trial.

Fluxes of N₂O started during the active stage of composting, occurring during the same time as CO₂ and CH₄ (Figure 3.4). On October 28 (DOY 300) there was a large spike of N₂O, at 1.10 g N m⁻² d⁻¹, immediately after the compost was turned. Frequent but variable occurrences of N₂O fluxes started on November 16 (DOY 319) and continued after the active stage of composting until December 19 (DOY 352); lasting 33 days in which daily fluxes ranged from 0 to 0.92 g N m⁻² d⁻¹.

At the start of the composting process NO fluxes began with a peak of 0.15 g N m⁻² d⁻¹ on September 28 (DOY 270) and continued until October 5 (DOY 277) when water was added (Figure 3.4). Fluxes were gap filled from September 23 to 28 (DOY 265 to 270); therefore, because NO fluxes were high on September 28 (DOY 270), fluxes may have been incorrectly estimated during this time. There is also a spike of NO flux, at 0.12 g N m⁻² d⁻¹, during the active stage of composting, on October 28 (DOY 300) along with N₂O, CH₄, and CO₂. After this NO flux were negligible.

Ammonia fluxes also began at the start of the composting process with a peak on September 28 (DOY 270) at 0.72 g N m⁻² d⁻¹ (Figure 3.4). Ammonia fluxes were gap filled from

September 23 to 28 (DOY 265 to 270) and therefore could have been inaccurately estimated during this time since NH₃ fluxes peaked on September 28 (DOY 270). Another spike of NH₃ occurred during the active stage on November 1 (DOY 304) at 0.89 g N m⁻² d⁻¹ and then declined shortly thereafter. Interestingly, this spike of NH₃ flux did not occur until 5 days after the October 27 (DOY 299) turning of the windrow; therefore, came slightly after the peaks of NO, N₂O, CH₄, and CO₂. Gas fluxes of NO₂ were negligible or negative in value; a trend was not identified (Figure 3.4).

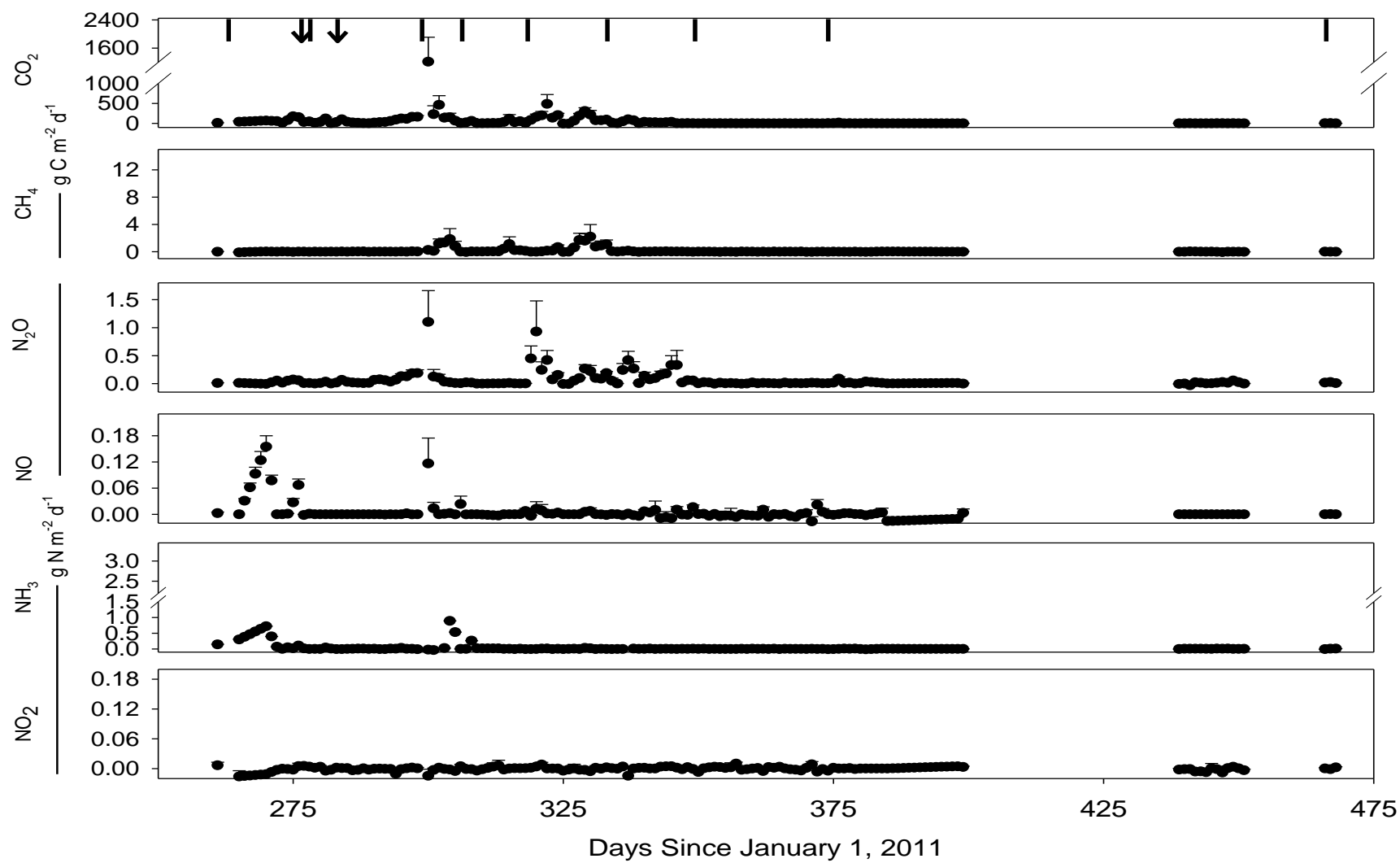


Figure 3.4 Trial 1 daily mean gas fluxes and where shown ± 1 standard error from September 14, 2011(DOY 257) to April 13, 2012 (DOY 468). Bars and arrows indicate windrow turning and water added, respectively. Data gap filling was used for total of 21 days.

3.4.3.2 Trial 2 Daily Mean Gas Fluxes

In Trial 2, the highest daily CO₂ flux was 1,750 g C m⁻² d⁻¹ on October 28 (DOY 300); the day after Trial 2 windrow was initiated (Figure 3.5). Fluxes of CO₂ gradually declined afterwards with slight spikes reaching around 500 g C m⁻² d⁻¹ each time the windrow was turned until January 10 (DOY 374) where afterwards CO₂ fluxes remained less than 10 g C m⁻² d⁻¹. The duration of active stage composting, as indicated by high CO₂ fluxes, lasted 74 days and ranged from 0 to 1,750 g C m⁻² d⁻¹ with an average of 123 g C m⁻² d⁻¹.

The production and release of CH₄ fluxes occurred the first day the windrow was initiated and followed a similar pattern to that of CO₂. The highest daily value of CH₄ flux was recorded at 10.46 g C m⁻² d⁻¹ on November 3 (DOY 306); which was immediately after the first time the windrow was turned. The compost windrow was turned again on November 15 (DOY 318), but CH₄ fluxes did not peak at 0.84 g C m⁻² d⁻¹ until four days later, on November 19 (DOY 322). The pattern of a delayed increase in CH₄ fluxes after turning the windrow repeated when the windrow was turned on November 30 (DOY 333) with CH₄ fluxes peaking at 3.48 g C m⁻² d⁻¹ on December 4 (DOY 337) and a peak of 1.39 g C m⁻² d⁻¹ on December 19 (DOY 352), three days after turning the windrow on December 16 (DOY 349). CH₄ fluxes reduced and were negligible after December 30 (DOY 363).

The release of N₂O fluxes were negligible until December 4 (DOY 337) when there was a slight increase to 0.14 g N m⁻² d⁻¹, four days after turning the windrow on November 30 (DOY 333). Fluxes of N₂O were also measured during the active composting stage on a regular daily basis, they started on December 17 (DOY 350) and continued past the active composting stage to January 15 (DOY 379); lasting a duration of 29 days, in which daily fluxes ranged from 0.02 to

0.58 g N m⁻² d⁻¹. In the spring, immediately after turning the windrow on April 11 (DOY 466), there were two days of N₂O fluxes, which reached a maximum of 0.32 g N m⁻² d⁻¹.

Fluxes of NO were highest at the beginning of Trial 2, starting at 0.09 g N m⁻² d⁻¹ on October 28 (DOY 300) and then declining by October 31 (DOY 303). Starting on December 4 (DOY 337) there was a two-day occurrence of NO fluxes, reaching 0.04 g N m⁻² d⁻¹. There were also some variable NO fluxes corresponding with N₂O fluxes from December 17 (DOY 350) to January 7 (DOY 371), as well as in the spring after turning the compost.

NH₃ fluxes started immediately upon initiation of Trial 2, with the highest peak of 2.25 g N m⁻² d⁻¹ on October 29 (DOY 301); however, there was a large amount of variability between chambers and declined shortly after by November 6 (DOY 309). On December 4 and 5 (DOY 337 and 338) NH₃ fluxes spiked at 0.31 and 0.88 g N m⁻² d⁻¹, respectively, and then became negligible thereafter. On December 4 (DOY 337), there was a simultaneous occurrence of CO₂, CH₄, N₂O, NO, and NH₃ peaks, four days after the windrow was turned. Gas fluxes of NO₂ were negligible or negative in value; no trends were identified (Figure 3.5).

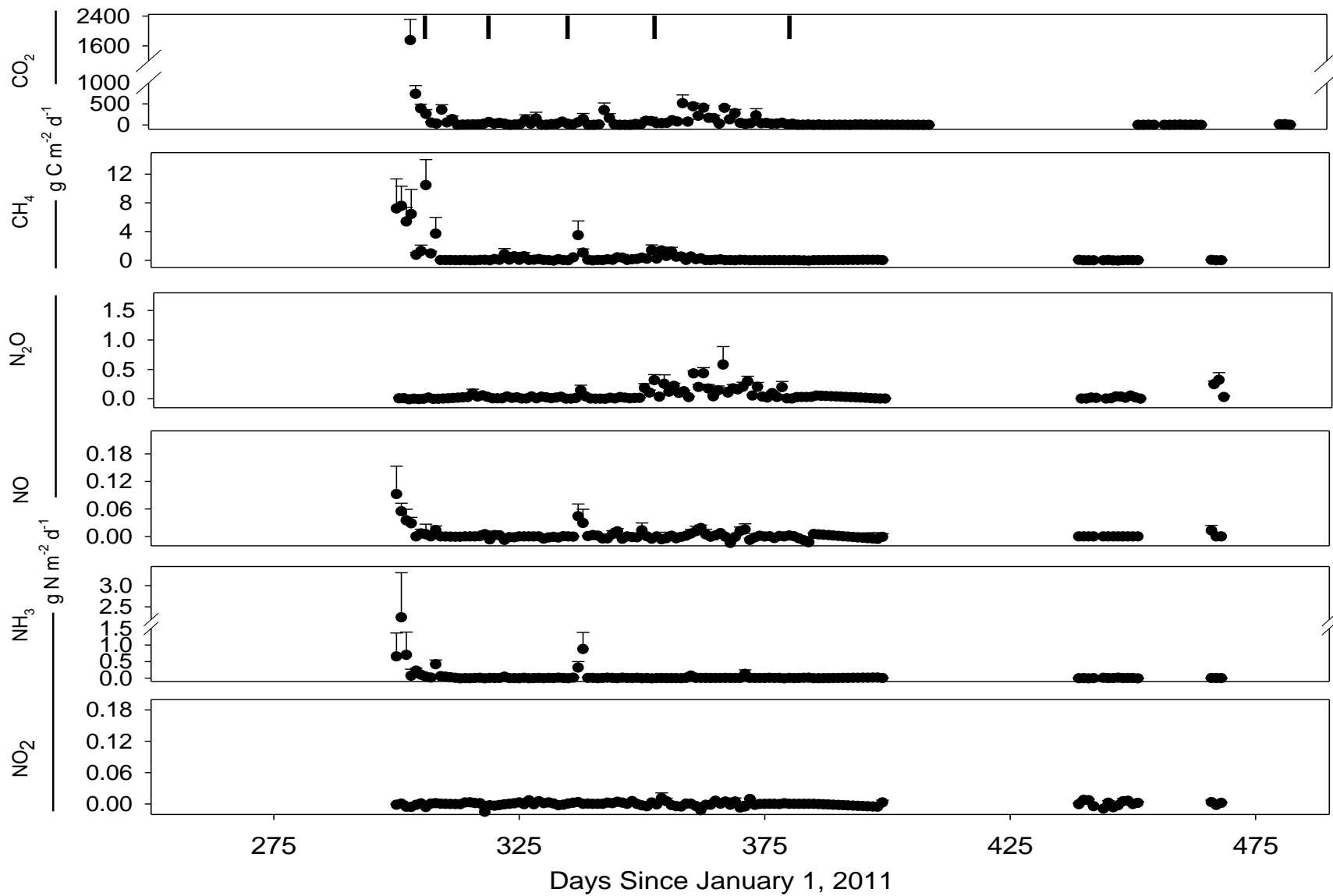


Figure 3.5 Trial 2 daily mean gas fluxes and where shown ± 1 standard error from September 14, 2011 (SOY 257) to April 13, 2012 (DOY 468). Bars indicate windrow turning. Data gap filling was used for a total of 5 days.

3.4.3.3 Cumulative Gas Emissions, Mass Balance and Greenhouse Gas Equivalent Values

Cumulative CO₂ emissions of Trial 1 were lower on a per surface area basis compared to Trial 2, 7,679 g C m⁻² and 9,363 g C m⁻² respectively, despite Trial 1 gas fluxes being measured for 207 days whereas in Trial 2 they were only measured for 168 days (Table 3.3). Trial 2 cumulative CO₂ emissions were also higher compared to Trial 1 when expressed as the total amount emitted, 970,868 g C compared to 531,086 g C, and total amount emitted per initial dry mass of the windrow, 197 g C Mg⁻¹ compared to 93 g C Mg⁻¹, respectively (Table 3.3 and 3.4). Cumulative CH₄ emissions were higher for Trial 2, at 1.49 g C Mg⁻¹ compared to Trial 1, at 0.16 g C Mg⁻¹ (Table 3.4). In Trial 1 and 2, there was considerable mass loss on a dry mass basis, losing 76% and 78% of the initial mass respectively, whereas Trial 3 only lost 23% of the initial mass on a dry mass basis (Table 3.4). Trial 1 and 2 had the same percentage of total C loss, losing 87.5% of the initial total C, which was greater than the 50.6% total C loss of initial total C in Trial 3. In Trial 1 and 2, the majority, over 99% of the measured cumulative C gases emitted, were from the emission of CO₂, with very little, less than 1%, produced by CH₄. However, only 23% and 54%, of the total C loss calculated by mass balance were accounted for by the measured emissions of CO₂ and CH₄ in Trial 1 and 2, respectively.

Total cumulative gaseous N emissions measured from Trial 1 were slightly lower than Trial 2 at 1,218 g N and 1,282 g N, respectively (Table 3.3). The majority of cumulative gaseous N emissions from both Trial 1 and 2 were a result of NH₃ emissions accounting for 46% and 58% of the total measured gaseous N emitted; N₂O was the second largest contributor with 46% and 39%, followed by NO with 8% and 3% (Table 3.4). Both Trial 1 and 2 had considerable total N losses calculated by mass balance at 12.9 g Mg⁻¹ and 12.5 g Mg⁻¹ which results in 70.4% and 73.7% N loss of initial total N (Table 3.4); whereas Trial 3 had a negative value for N loss calculated by

mass balance, -2.0 g Mg^{-1} . However, cumulative gaseous emissions measured for Trial 1 of N_2O , NO , and NH_3 were 0.1 g N Mg^{-1} , 0.02 g N Mg^{-1} , and 0.15 g N Mg^{-1} , respectively, therefore only 1.7 % of the total N losses calculated by mass balance was accounted for by the cumulative N emissions measured (Table 3.4). Similarly, in Trial 2, cumulative gaseous emissions measured for N_2O , NO , and NH_3 were 0.1 g N Mg^{-1} , 0.2 g N Mg^{-1} , and 0.15 g N Mg^{-1} , respectively, accounting for only 2.1 % of the total N losses calculated by mass balance (Table 3.4). Trial 1 and 2 had similar cumulative N_2O emissions measured, Trial 1 had higher cumulative NO emissions measured, and Trial 2 had higher cumulative NH_3 emissions measured based on values per initial dry mass (Table 3.4). Values for cumulative NH_3 emissions measured are higher than N_2O in total g N and per initial dry mass basis but not per surface area because the surface area of the windrow was larger at the beginning of the composting process when NH_3 fluxes were occurring and smaller when N_2O fluxes were occurring (Table 3.3 and 3.4).

Comparing direct greenhouse gas emissions, considering only cumulative emissions of CH_4 and N_2O using global warming potential (GWP) factors of 28 for CH_4 and 265 for N_2O , cumulative emissions expressed as CO_2 equivalents were 46.5 kg Mg^{-1} for Trial 1 compared to 97.9 kg Mg^{-1} for Trial 2 (IPCC 2013: GWP100 without inclusion of climate-carbon feedbacks) (Myhre et al., 2013). In Trial 1, 13% of the GWP was from CH_4 emissions and 87% was from N_2O emissions. In Trial 2, 57% was from CH_4 emissions and 43% was from N_2O emissions.

Table 3.3 Cumulative gas emissions measured based on surface area and total amount measured as emissions from Trial 1 and 2.

Gas	Trial 1	Trial 2	Trial 1	Trial 2
	g C or N m ⁻²		g C or N	
CO ₂	7,679	9,363	531,086	970,868
CH ₄	17.68	60.97	896	7,363
N ₂ O	9.39	7.25	559	500
NO	0.66	0.33	93	40
NH ₃	4.43	5.83	566	742
NO ₂	-0.09	-0.04	-12	-3

Cumulative emissions were calculated based on a span of 207 and 168 days for Trial 1 and 2, respectively. Total emissions measured (g C or N) were calculated based on estimated surface area at the time of emission.

Table 3.4 Initial, final and losses of mass, total carbon (TC) and total nitrogen (TN) calculated by mass balance expressed as initial mass of the windrows as well as cumulative gas emissions measured expressed as initial mass of windrow. Losses of mass, TC, and TN are expressed as a percentage of initial mass, initial TC, or initial TN respectively as well as cumulative gas emissions expressed as percentage of the initial TC or initial TN, as a percentage of the TC or TN losses, and as a percentage of total C or N gaseous emissions measured for Trial 1, 2, and 3.

	Mass	TC	TN	CO ₂	CH ₄	N ₂ O	NO	NH ₃
	kg Mg ⁻¹			kg C or N Mg ⁻¹				
Trial 1								
Initial	1,000	453	18.3					
Final	242	57	5.4					
Total Loss or Emission	758	396	12.9	93	0.16	0.10	0.02	0.10
				Percent (%)				
Of Initial Mass, TC, or TN	75.8	87.5	70.4	20.5	0.0	0.5	0.1	0.5
Of Total C or N Losses				23.4	0.0	0.8	0.1	0.8
Of Total C or N Gaseous Emissions				99.8	0.2	45.8	7.7	46.5
Trial 2								
Initial	1,000	420	16.9					
Final	220	53	4.4					
Total Loss or Emission	780	367	12.5	197	1.49	0.10	0.01	0.15
				Percent (%)				
Of Initial Mass, TC, or TN	78.0	87.5	73.7	46.9	0.4	0.6	0.1	0.9
Of Total C or N Losses				53.7	0.4	0.8	0.1	1.2
Of Total C or N Gaseous Emissions				99.2	0.8	39.0	3.1	57.9
Trial 3								
Initial	1,000	369	14.5					
Final	769	182	16.5					
Total Loss or Emission	231	187	-2.0	N/A	N/A	N/A	N/A	N/A
				Percent (%)				
Of Initial Mass, TC, or TN	23.1	50.6	-13.6	N/A	N/A	N/A	N/A	N/A

Mass, total carbon (TC), and total nitrogen (TN) are on dry mass basis. Values expressed as Mg⁻¹ are based on initial dry mass of the windrow. NO₂ was not included in calculations because values were negative due to random variability.

3.4.4 Final Compost Composition

Maturity indicators, included C:N, $\text{NH}_4^+\text{-N}:\text{NO}_3^-/\text{NO}_2^-\text{-N}$, and final temperature of the windrow, were similar amongst the three Trials (Table 3.5). In all the Trials, the final C:N was 12 or less (Table 3.5). For the C:N there was statistical significance between the Trials; however, this was due to low variability amongst the samples. There were still straw pieces visible in all three Trials, Trial 3 had the most frequent and largest pieces, whereas Trial 1 had the fewest and smallest. The straw pieces were degraded and resembled fibres. Trial 2 had statistically significant lower $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-/\text{NO}_2^-\text{-N}$ values compared to Trial 1 and 3; however, the ratios of $\text{NH}_4^+\text{-N}:\text{NO}_3^-/\text{NO}_2^-\text{-N}$ for Trial 1, 2, and 3 were all similar at 2.4, 1.5, and 2.0, respectively (Table 3.5). The final temperature of the compost windrows was similar to ambient air temperature (Figure 3.2).

The final compost parameters of agronomic importance, including OM, Total N, P, K, pH, EC, Total Na, BD, and MC were also similar amongst the three Trials (Table 3.5). There were no statistical differences in OM, values ranged from 423 to 429 g kg^{-1} (Table 3.5). There were no statistical differences in Total N, values ranged from 20.2 to 22.4 g kg^{-1} (Table 3.5). It is important to note that the starting Total N values were 18.3, 16.9, 14.5 g kg^{-1} for Trials 1, 2, and 3, respectively, therefore in all windrows the Total N increased in concentration from the initial analysis compared to the final analysis. Trial 2 had statistically significant lower Total P compared to Trial 1 and 3, values ranged from 8.3 to 10.5 g kg^{-1} (Table 3.5). All Trials had statistically different Total K values, which ranged from 9.1 to 13.5 g kg^{-1} (Table 3.5). The standard error amongst the samples for pH, EC and Total Na was very small; there were statistically significant differences for pH and EC, but no statistical differences for Total Na (Table 3.5). The values for pH, EC, and Total Na ranged from 7.4 to 7.8, 1.6 to 2.0 mS cm^{-1} , and 1.1 to 1.2 g kg^{-1} , respectively. Each BD value was significantly different; the BD of Trial 3 was lowest at 401 kg m^{-3} , while Trial

1 was highest at 536 kg m^{-3} (Table 3.5). The BD values correlate with the length of time the feedstock material had been composting as well as the moisture content. The longer the feedstock material had been composting, the higher the BD, additionally, the higher the moisture content, the higher the BD. There was no statistical difference in MC between Trial 1 and 2, with MC at 567 and 550 g kg^{-1} , respectively, but Trial 3 was significantly lower with a MC of 504 g kg^{-1} (Table 3.5).



Figure 3.6 Samples of Trial 1, 2 and 3 windrows from the starting dates September 14 (DOY 256), October 27 (DOY 299), and November 15 (DOY318), 2011 and the final product from Trial 1, 2, and 3 on April 11, 2012 (DOY 466).

Table 3.5 Characteristics of Trial 1, 2, and 3 final compost analysis sampled on April 11, 2012 (DOY 466).

Trial	BD kg m ⁻³	C:N Ratio	pH	EC mS cm ⁻¹	MC	Total N	Total P	Total K	Total Na	Total OM	NH ₄ -N mg N kg ⁻¹	NO ₃ /NO ₂ -N mg N kg ⁻¹
1	536 (8) ^a	10 (0) ^b	7.8 (0) ^a	1.6 (0) ^b	567 (8) ^a	22.4 (1.0) ^a	10.3 (0.7) ^a	10.6 (0.4) ^b	1.2 (0.1) ^a	423 (12) ^a	1250 (78) ^a	531 (46) ^{ab}
2	445 (6) ^b	12 (0) ^a	7.7 (0) ^{ab}	2.0 (0) ^a	550 (3) ^a	20.2 (0.1) ^a	8.3 (0.1) ^b	13.5 (0.1) ^a	1.1 (0) ^a	429 (2) ^a	700 (15) ^b	465 (28) ^b
3	401 (15) ^c	11 (0) ^{ab}	7.4 (0) ^b	2.0 (0) ^a	504 (4) ^b	21.4 (0.2) ^a	10.5 (0.1) ^a	9.1 (0.1) ^c	1.2 (0) ^a	427 (4) ^a	1303 (43) ^a	654 (26) ^a

Values are the mean of three samples ± 1 standard error in parenthesis. Values for Bulk Density (BD) and Moisture Content (MC) are reported on an as-is basis and those for Total Nitrogen (N), Phosphorus (P), Potassium (K), Sodium (Na), Organic Matter (OM), Ammonium (NH₄-N), and Nitrate/Nitrite (NO₃/NO₂-N) on a dry mass basis. Means within the same column followed by the same lowercase letter (a, b, c) are not significantly different at P<0.05 ANOVA Tukey Test.

3.5 Discussion

3.5.1 Effect of Starting Date and Winter Conditions on the Process of Composting Pig Slurry Solids

Air temperature and precipitation had an effect on the moisture content and temperature of uncovered compost windrows. During the months of September and October 2011 temperatures were high, precipitation was low, and the open field was prone to high winds. Trial 1 lost moisture through evaporation during the first month after it was initiated due to warm air temperatures, wind, internal heat from the compost, and very little precipitation (Environment Canada, 2013). The loss of moisture reduced microbial activity, indicated by reduced temperatures and reduced CO₂ emissions, and required the addition of water to re-establish microbial activity. Covering the windrow with a breathable material or starting with a larger pile, with a smaller surface area to volume ratio, may have reduced water evaporation (Environment Canada, 2013). Maintaining moisture content in the optimal range not only keeps the composting process fluent, but this could prevent having to add water and risk leaching the slurry solids from the windrow.

In Trial 2, the warm, windy conditions, small pile size, and frequent turning assisted with the evaporation of the water, reducing the water content down to the optimal range, but these weather conditions cannot always be relied on (Environment Canada, 2013). Starting a windrow with feedstock materials above the desired range of moisture content will create anaerobic conditions (Environment Canada, 2013). With straw and slurry solids as feedstock materials recovering from the initial high moisture content was difficult. The extra moisture weakened the structure of the straw and liquefied the slurry solids, which collapsed and compacted the pore space, making it difficult to provide passive airflow through the windrow. Anaerobic microbial organisms tend to generate less heat and reduce the rate of decomposition, which was not ideal for

composting, but especially not ideal in winter conditions to maintain thermophilic temperatures (Environment Canada, 2013). The moisture content for windrows started in or near winter should be within the optimal range or slightly below because the addition of snow can maintain or increase the moisture content, as seen in Trial 3 as well as in Larney et al. (2000) and McCartney and Eftoda (2005). Incorporating the accumulated snow into the windrow by turning increased the moisture content and the cool air temperatures reduced evaporation.

Despite winter conditions having an effect on the composting process, it was possible to reach thermophilic temperatures while composting slurry solids with straw. The thermophilic period, 59-73 days, was shorter compared to 132 days documented by Larney et al. (2000) while composting feedlot manure or 98 days for composting municipal biosolids during winter conditions (McCarthy and Eftoda, 2005). The time requirement of the CCME (2005) Compost Quality Guidelines PFRP recommendations for killing pathogens was met in all three trials; however, the number of turns required, 5 times while temperatures are above 55 °C, was not met for Trial 2 or 3. Windrow turning was required to provide oxygen to organisms, expose all feedstock material to thermophilic temperatures, mix nutrients, and restructure pore spaces (Environment Canada, 2013). However, turning also introduced cold air and released heat which slowed microbial activity and caused a lag in the time it took for the windrow to heat up again or the windrow did not heat up again at all (Environment Canada, 2013; McCartney and Eftoda, 2005). In all three trials, turning the windrows while thermophilic conditions were occurring did not prevent the windrow from heating up again. During the thermophilic stage of Trial 2 and 3, the frequency of turning could have been increased to ensure the PFRP requirement could have been met. Waiting to turn the windrow when mesothermic temperatures were occurring in Trial 1 and 2 and the air temperature was -10 °C resulted in an abrupt decline in the compost temperature

and the microbial population activity. McCartney and Eftoda (2005) also could not reach thermophilic conditions after turning in cold air conditions when windrow temperatures were low. The abrupt decline in activity may have prematurely started the curing phase. In all trials there were pieces of straw visible in the final product, despite displaying other maturity indicators by April 2012. The appearance of the final product would be suitable for an agriculture application but may require screening and blending with topsoil to be used in landscaping or horticultural commercial use.

Turning compost windrows in the winter with a Brown Bear compost turner required extra caution to maintain the compost pad if working on a soil surface. The unevenness of the snow made it easy for the blade of the Brown Bear compost turner to dig into the compost pad; this added clay soil to feedstock material; which made it difficult to turn in the future. Feedstock material stuck to the clay soil and it became hard clumps of clay with pockets of anaerobic odorous material. These clumps of clay did not provide the expected crumbly texture of compost.

3.5.2 Greenhouse Gas and Nitrogen Gaseous Losses with Respect to Composting Conditions

As expected, CO₂ fluxes occurred throughout the composting process and the flux rate responded to conditions and management of the windrow. In general, CO₂ fluxes were highest the first few days immediately after turning the windrows. Turning allowed mixing and re-structuring of the materials for the microbial population to access nutrients and replenish oxygen in the pore spaces allowing for rapid microbial activity and CO₂ production during this time. The compost temperature and CO₂ rates were closely associated. Studies consistently show CO₂ emissions highest at the beginning of the composting period (Hao et al 2001; Hao et al 2011; Paillat et al., 2005; Maeda et al., 2010). However, at the beginning of the composting process in Trial 1, CO₂

fluxes were comparatively low and it was observed that the wind speed was high. Wind blowing horizontally through the windrow can reduce the amount of gas fluxes being captured in the chamber on top of the windrow. Carbon dioxide fluxes and wind speed were compared for September 19 (DOY 261) to October (DOY 299) against the remaining active stage and it was determined that wind speed was no higher during this time compared to the rest of the active period (see Chapter 5). Therefore, high winds may have been a factor in reducing CO₂ fluxes overall but CO₂ fluxes at the start of the composting process were lower compared to the rest of the active stage; likely due to low moisture content inhibiting the activity of the microorganisms (Environment Canada, 2013). The maximum daily CO₂ flux, 600 g C m⁻² d⁻¹, measured by Hao et al (2004) using static vented chambers while composting cattle feedlot manure, was lower than the maximum peak of daily CO₂ flux measured using the FTIR and automated chamber system, 1,212 g C m⁻² d⁻¹ and 1,750 g C m⁻² d⁻¹, for Trial 1 and 2 respectively. Despite measuring gases from Trial 1 for a longer period of time compared to Trial 2, cumulative CO₂ emissions were higher in Trial 2 as a result of a longer active stage and higher daily flux average during this time. Cumulative CO₂ emissions for Trial 1, 93 kg C Mg⁻¹ (initial dry mass), was lower than the 145 to 165 kg C Mg⁻¹ range measured by Hao et al. (2004); whereas the cumulative CO₂ emissions of Trial 2 was above at 197 kg C Mg⁻¹. However, in this study there was considerable total mass loss in Trial 1 and 2, over 75 % which is related to total carbon loss, both 88 %, compared to the 27-30 % total mass loss and 35-53 % total carbon loss reported by Hao et al. (2004) after 99 days of composting. This is important because only 23.4 % and 53.7 % of the total estimated C loss was accounted for by CO₂ emissions in Trial 1 and 2, whereas Hao et al. (2004) accounted for 95 % of the total C loss through CO₂ emissions. The study by Hao et al. (2004) was conducted in a semi controlled environment with a roof and concrete floor, whereas this study was prone to C losses

by runoff or leaching as well as substantial removal of composting material through sampling and turning equipment, the total amount removed was not recorded. Additionally, the final mass was estimated based on volume and bulk density instead of being weighed. It was also assumed that the MC, BD, and Total C that were measured from samples were representative of the entire windrow. Therefore, unaccounted losses of mass and Total C could also be a result of other factors in addition to underestimation of CO₂ emissions. In order to conduct mass balance analysis, the environmental and management conditions should be controlled in order to minimize unaccounted for losses, and the initial and final material should be weighed with a scale instead of through estimation of the volume and bulk density.

As expected, CH₄ fluxes corresponded with the amount of oxygen present, the availability of carbon, and if there was an emission pathway (Hao et al., 2001; Ahn et al., 2011; Fukumoto et al., 2003; Sommer and Moller 2000; Jiang et al., 2011). At the beginning of Trial 1 the moisture content and microbial activity, based on CO₂ emissions, were low, this resulted in adequate oxygen content. Conditions were not present for producing CH₄, however, turning the windrow allowed for the release of the small amount of CH₄ that was produced in small pockets of anaerobic zones or near the base of the windrow. Prior to the other two peaks of CH₄ during Trial 1, there was a gradually increased in flux rate, it appeared that there was a gradual build-up of CH₄ that was then slowly released when the concentration produced exceeded that of which could be oxidized by methanotrophic bacteria (Hao et al., 2001; Ahn et al., 2011). The peaks also correspond to the increase in CO₂ emission; therefore, lack of air due to rapid microbial activity was likely the cause of anaerobic conditions as opposed to excess moisture displacing oxygen. Unlike Trial 1, the moisture content of Trial 2 was high at the start of the composting process; which displaced oxygen and created conditions for CH₄ production and emission from the windrows. It was likely that CH₄

fluxes were underestimated because they were not captured during the first turning of the windrow because the chambers had to be removed. The other CH₄ peaks were similar to that seen in Trial 1, a gradual increase in flux rate as CH₄ concentration built up within, indicating reduced oxygen due to rapid microbial activity instead oxygen being displaced by moisture. After the available carbon had decreased, indicated by the reduced C:N, and there wasn't a demand for oxygen CH₄ fluxes were no longer observed. These mechanisms of CH₄ production and emissions have been observed and are similar to those discussed in previous studies (Hao et al., 2001; Ahn et al., 2011; Fukumoto et al., 2003; Sommer and Moller 2000; Jiang et al., 2011). The majority of CH₄ fluxes in Trial 2 came from the initial 10 days of the composting process when the initial moisture content was above the optimal range. This led to higher cumulative CH₄ emissions in Trial 2 compared to Trial 1. However, the maximum daily CH₄ flux, 80 g C m⁻² d⁻¹, measured by Hao et al. (2004) was much higher, than the maximum peak of daily CH₄ flux measured in Trial 1 and 2 at 1.85 g C m⁻² d⁻¹ and 10.46 g C m⁻² d⁻¹, respectively. Additionally, this study measured total cumulative CH₄ emissions as 0.16 and 1.49 kg C Mg⁻¹, Trial 1 and 2, whereas Hao et al. (2004) reported cumulative CH₄ emissions as high as 8.93 kg C Mg⁻¹. Accordingly, Hao et al (2004) also reported CH₄ emissions representing up to 5.8 % of total C loss and 2.7 % of initial total C, whereas in this study CH₄ emissions accounted for less than 0.5 % of total C loss and 0.05 % of initial total C. Anaerobic conditions were observed in this study, through measurement of low oxygen concentration and high moisture content, to be similar as those observed in Hao et al. (2004). Therefore, since the values in this study are lower than Hao et al (2004) and there was substantial unaccounted for C loss, there is potential that substantial CH₄ fluxes were not measured during this study.

In addition to substantial mass and total C losses in Trial 1 and 2 in this study, over 70 % of the total N was lost in Trial 1 and 2 as well. In comparison, Hao et al. (2004) reported total N

losses between 11-42 %; however, that study also indicated that N₂O emissions measured only accounted for 0.9-5.9 % of the total N loss, and suggested the remainder were potentially NH₃ or N₂ emissions that were not measured. In this study, N₂O, NH₃, and NO gaseous emissions still only accounted for less than 2 % of Total N loss calculated by mass balance; therefore, this suggested that substantial gaseous losses may not have been captured, there could have been N losses through leaching, runoff, and removal of material through sampling and equipment, and the estimated mass and samples of MC, BD, or Total N for calculating Total N losses may not have been representative of the windrow.

In this study the presence of N₂O fluxes could have occurred as a result of both identified pathways, denitrification and incomplete nitrification, as described by Hao et al. (2004), Jiang et al. (2011), Fukumoto and Inubushi (2009), and Fukumoto (2012). Fluxes of N₂O started midway through the active stage of composting, occurring at the same time as CO₂ and CH₄, shortly after the windrows were turned. During this time NO₃⁻/NO₂⁻-N had begun to increase in concentration and there were anaerobic conditions within the center of the windrows. Nitrification could have occurred on the surface of the windrow and produced NO₃⁻, when the windrow was turned the NO₃⁻ was exposed to anaerobic conditions and denitrification could have produced N₂O (Fukumoto and Inubushi, 2009; Fukumoto, 2012). The continuation of N₂O fluxes, after CO₂ and CH₄ fluxes and compost temperature declined, could have been due to incomplete nitrification, observed and described in other studies as well (Hao et al., 2001; Hao et al., 2004; Fukumoto and Inubushi, 2009; Fukumoto, 2012; Ahn et al., 2009). The maximum daily N₂O flux, 0.7 g N m⁻² d⁻¹, measured by Hao et al (2004), was lower than the maximum peak of daily N₂O flux measured in Trial 1 at 1.10 g N m⁻² d⁻¹, but within that range when N₂O fluxes were occurring on a frequent basis for Trial 1 and 2. Cumulative emissions of N₂O were comparable between the two trials

reporting $0.10 \text{ kg N Mg}^{-1}$, which is comparable to Hao et al. (2004) reporting up to $0.08 \text{ kg N Mg}^{-1}$. Additionally, in this study N_2O emissions accounted for up to 0.6 % initial N and 0.8 % total N loss. This is within the range of N_2O emissions accounting for 0.39-0.68 % initial N loss and 0.9-5.9 % of total N loss reported by Hao et al. (2004), since similar nitrogen content and windrow management was utilized these relatively modest N_2O emissions were expected. Other studies have observed higher N_2O losses, in the range of 1.0-9.27 % of initial total N either due to increased forced aeration or turning, that allows either increased nitrification and diffusion of N_2O or turning when there were anaerobic conditions (Jiang et al., 2011; Ahn et al., 2011; Hao et al., 2001; Fukumoto et al., 2003; Fukumoto and Inubushi, 2009). When gaseous emissions from composting are measured for an extended period of time it becomes evident that N_2O losses can be as substantial as NH_3 . In this study N_2O accounted for a third to nearly half the total amount of gaseous N emissions measured, similar to the findings in Fukumoto and Inubushi (2009).

Fluxes of NH_3 occurred at the very start of the composting process corresponded with high levels of NH_4^+ . Even though the average pH was not high enough to encourage NH_4^+ conversion to NH_3 , there were likely isolated areas where conditions existed for NH_3 production, especially in Trial 2 since the average pH was quite high (Environment Canada, 2013). The C:N was within the optimal starting range but the C in the straw may not have been available to stabilize the readily available N or there was not good contact between the straw and slurry solids (Environment Canada, 2013). Gap filling was done at the very start of Trial 1, between September 23 to 28 (DOY 265 to 270), a very critical time for potential NH_3 fluxes, therefore NH_3 could have been over or underestimated during this time. Initial NH_3 fluxes in Trial 2 had greater variability between chambers compared to Trial 1. In both Trials there was a flux of NH_3 a few days after turning the compost during the middle of the active stage of composting, but not after every turning event.

Overall, cumulative NH_3 emissions were slightly higher in Trial 2 compared to Trial 1, this is surprising since Trial 1 had higher $\text{NH}_4^+\text{-N}$ to begin with, had lower initial moisture content, larger pore spaces allowing for pathways for NH_3 to be emitted from, and was turned more frequently during the active stage; which are all factors for increasing NH_3 losses (Larney et al., 2006; Hao and Chang, 2001; Fukumoto et al., 2003; Fukumoto and Inubushi, 2009; Maeda et al., 2009; Jiang et al., 2011; Paillat et al., 2005; Elwell et al., 2002). In this study cumulative NH_3 emissions represented between 0.54-0.89 % of the initial total N, whereas previous studies have reported NH_3 emissions to be anywhere from 0.3 % of the initial N up to 34 % (Fukumoto and Inubushi, 2009; Hansen et al., 2006; Jiang et al., 2011). Therefore, it was reasonable to have expected higher NH_3 emissions, knowing that there was not continuous flux measurement at the beginning of Trial 1 and considerable variability in measurements in Trial 2, there was potential that NH_3 fluxes were not accurately captured during this study.

Fluxes of NO occurred at the beginning of the composting process, corresponding with NH_3 fluxes, this has been explained by NH_4^+ undergoing rapid nitrification and producing NO when oxygen was present (Hao and Chang, 2001; Fukumoto, 2012). Once again, gap filling during the start of the composting process may have over or underestimated NO in Trial 1. Fluxes of NO also occurred with N_2O after the windrows were turned during the active stage of composting, indicating NO production through denitrification (Hao and Chang, 2001; Fukumoto, 2012). Cumulative emissions of NO were slightly higher in Trial 1 compared to Trial 2; this was expected as Fukumoto (2012) noted that in dry conditions, such as near the start of Trial 1, NO fluxes can be higher than if moisture conditions are high. Fukumoto (2012) has stated NO emissions can be anywhere from one tenth of those for N_2O up to half if increasingly dry conditions existed. This suggests that the amount of NO measured in this study was expected. Fluxes of NO_2 were not

recorded, however, since NO fluxes were measured, it was likely that NO oxidized to NO₂ in the atmosphere but was not captured using the chambers (Hao et al 2001).

In Trial 3, the total N loss was calculated by mass balance was -2.0 kg Mg⁻¹; indicating N losses were less than dry mass loss (Larney et al., 2006). If the study was carried on into the summer it would have been interesting to see if mass, carbon, and N losses increased. It was unfortunate gaseous emissions were not measured during Trial 3 to compare the conditions present and management of the windrows to the other Trials. Since the total cumulative N gaseous losses measured were similar in both Trial 1 and 2, 0.22 g N Mg⁻¹ and 0.26 g N Mg⁻¹, respectively, there was nothing in particular about differences in conditions present or management of the windrows that could be suggested to reduce N gaseous losses.

In terms of greenhouse gas emissions, CO₂-equivalent values for CH₄ and N₂O in Trial 1 were less compared to Trial 2 at 46.5 kg CO₂-equivalent Mg⁻¹ and 97.9 kg CO₂-equivalent Mg⁻¹, respectively. The anaerobic conditions in Trial 2, that produced greater CH₄, did reflect that greenhouse gas emissions could be extensive. It was expected for CH₄ to have a larger contribution to CO₂-equivalent values, despite having a lower GWP compared to N₂O (Hao et al., 2004; Zhong et al., 2013). However, in comparison to other studies, the values calculated were low, Jiang et al (2011) measured CH₄ and N₂O contributing an average of 533 kg CO₂-equivalent Mg⁻¹ and Hao et al. (2004) measured 177 kg CO₂-equivalent Mg⁻¹. This further suggests CH₄ losses may not have been accounted for in this study.

3.5.3 Overall Compost Quality

The parameters measured indicate high quality compost produced from slurry solids and straw. The compost produced in each of the Trials show indicators of a mature product. The temperature

of the windrows in all Trials did not re-heat above ambient temperatures in the spring after the windrows were turned, indicating low microbial activity. Additionally, the C:N in the final compost product for each of the Trials was below 20, therefore the possibility of tying up plant available nitrogen when it is applied is minimal (Larney et al., 2006). However, it was expected that the C:N would be higher since straw pieces remained visible in all Trials. In Trial 1, frequent analysis allowed observation of a rapid decline in C:N during the active stage of composting and then remained stable. This demonstrates why C:N should not be the only indicator of maturity. Another indicator of maturity is the ratio of $\text{NH}_4^+\text{-N}:\text{NO}_3^-/\text{NO}_2^-\text{-N}$ with values less than 10 indicating a stable product (Bernal et al. 1998). All three Trials also met this indicator with values closer to 2.5.

The compost produced also had agronomic values that indicate high quality. Typically, Total N concentration in final compost is between 8-16 g kg⁻¹, Total P is between 1.3-4.4 g kg⁻¹ and Total K is between 4.1-12.4 g kg⁻¹ (Paul and Geesing, 2009). The compost produced from each Trial have Total N and P concentrations above the typical values and Total K within the ideal range, there was also little variability in these values between the Trials. Additionally, in all the Trials the total N values increased in concentration from the initial sampling compared to the final sampling. The increase in concentration of Total N indicate that losses of N through volatilization, leaching, or runoff, were less than the rate of decomposition or removal of material, indicated by mass loss (Larney et al. 2006).

The preferred range of pH is between 6.5 and 8, outside this range there was potential for mobilizing certain trace elements (Paul and Geesing, 2009). The pH of the compost produced in all Trials finished within this range. Electrical conductivity, measures the soluble salts including salts that provide nutrients such as potassium, magnesium, calcium, nitrates and ammonia as well

as salts that can be toxic to plant roots such as sodium and chloride. Epstein (1997) suggests anything greater than 5 mS cm^{-1} may lead to phytotoxicity, therefore soluble salts should not be a concern for using the compost produced from these Trials; however as with all compost it should be incorporated into soil. The total Na was reported to present the relative values for each Trial. There is no known recommended value or limit for total Na because toxicity of Na was dependent on solubility and the solubility of Na in compost was not known.

Other agronomic parameters such as BD and moisture content are helpful in determining the ease of use of the finished compost. The BD was important if it needs to be used to estimate mass for transporting. Typical BD for compost can be between $557\text{-}872 \text{ kg m}^{-3}$ (Paul and Geesing, 2009), the BD in all 3 Trials were slightly below this range. Since BD was measured as is, knowing the moisture content was important, higher moisture content can increase the BD. Moisture content can be reduced to 400 g kg^{-1} to reduce the mass of transporting the compost while still maintaining the microbial populations (Environment Canada, 2013). The moisture content was high in all three Trials at the time of sampling; turning the compost or giving it more time to dry out can reduce moisture. The organic matter of the compost produced was in the mid to low range typical of compost which can be anywhere from $300\text{-}700 \text{ g kg}^{-1}$ (Paul and Geesing, 2009). The addition of the clay soil from the composting pad surface may have introduced inorganic material into the compost, therefore lowering these values. Only a few parameters showed statistical significance between the Trials, those that did were due to low variability in the sampled product. This indicates consistency of end product and the number of samples taken of the final product could be reduced.

3.6 Conclusion

This study determined that temperature requirements in order to meet PFRP criteria were possible when composting slurry solids throughout the winter in Manitoba. When the windrow was turned in cold air temperatures there was a lag time before thermophilic temperatures are reached again. Turning the windrow in the winter once mesothermic temperatures had begun prematurely started the curing phase of the composting process because the microorganisms could not re-generate enough heat. In practice, a producer may want to consider combining windrows or re-structure the windrow to reduce the surface area to volume ratio after the active stage of composting to provide better insulation from winter conditions. Since the windrows were not sheltered from wind or precipitation, seasonal changes impacted the moisture content of the windrows, which if too high or low, can result in reduced temperature. Windrows must be managed in order to meet the five turns required for PFRP criteria, while also considering moisture content, compost temperature, and air temperature. Trial 1 was able to meet the PRFP criteria, but Trial 2 and 3 needed to be turned more frequently at the beginning of the composting process in order to meet the PFRP criteria.

Conditions did exist that allowed the emission of greenhouse and nitrogen gases. The frequency of the data collection made it possible to identify the different processes involved throughout the composting process and can assist in determining best management practices. However, it was important that the monitoring is continuous, especially during potentially critical times such as at the beginning of the composting process to capture NH_3 and immediately after turning to capture CH_4 . Trial 2 emitted more direct greenhouse gas emissions compared to Trial 1, 97.9 kg CO_2 -equivalent Mg^{-1} and 46.5 kg CO_2 -equivalent Mg^{-1} , respectively, due to the higher moisture content at the start of Trial 2. There was very little difference between Trial 1 and 2 in

measured N gaseous losses with a total of 0.22 g N Mg⁻¹ (N₂O 0.10 g N Mg⁻¹, NO 0.02 g N Mg⁻¹, and NH₃ 0.10 g Mg⁻¹) and 0.26 g N Mg⁻¹ (N₂O 0.10 g N Mg⁻¹, NO 0.01 g N Mg⁻¹, and NH₃ 0.15 g Mg⁻¹), respectively. The timing and quantity of these greenhouse and gaseous losses were comparable to other studies and could be related to the properties and management of the windrows. Gaseous emissions measured did not account for Total C and N losses calculated by mass balance. This disparity may have been a result of underestimated gaseous emissions measured because of high winds as well as overestimation of Total C and Total N losses calculated by mass balance due to environmental factors such as runoff, leaching, and the removal of material through sampling and on equipment that were not accounted for.

The compost produced from all three trials was a marketable product based on maturity and agronomic parameters. Despite the remainder of visible straw pieces, the other maturity indicators, temperature, C:N and NH₄⁺-N:NO₃⁻/NO₂⁻-N, indicated the product was mature. The agronomic parameters including nutrients, pH and EC, and ease of use, value this as a high quality product. Nutrient values of Total N, P, and K were above typical values seen in compost. There were no concerns with the pH and the amount of soluble salts should not be a concern if the compost was incorporated in the soil. In all three Trials the moisture content could be lowered in order to decrease the mass for efficient transportation and application. There was very little variability in the parameters measured in the end product, the differences in management did not affect the outcome of the product, if similar starting materials are used a consistent product can be produced.

Considering that temperatures required for PFRP can be met and a consistent quality product can be produced, composting was a suitable option for managing slurry solids on-farm throughout the winter and could be recommended to producers. Additionally, the quantity and

timing of greenhouse and nitrogen gaseous emissions measured by the automated chambers and FTIR could be explained by the physical properties and management of the windrows; therefore, this was a tool that could be utilized for future studies that compare management practices.

4.0 EFFECT OF BULKING MATERIAL ON GREENHOUSE GAS AND NITROGEN GASEOUS LOSSES DURING COMPOSTING OF SEPARATED PIG SLURRY SOLIDS IN WINTER

4.1 Abstract

The evaluation of bulking material, straw or woodshavings, was conducted in order to make recommendations for best management practices for composting separated pig slurry solids. The bulking materials were compared based on the ability to provide insulation throughout the winter while minimizing greenhouse gas and nitrogen gaseous losses during the composting process, and produce a compost product that meets CCME Guidelines for Compost Quality criteria including maturity, presence of foreign materials, thresholds for trace element concentrations, and pathogen levels as well as have ideal agronomic parameters in order to use or market the product. Both types of bulking materials were able to provide conditions for thermophilic microbial activity and insulation to maintain thermophilic conditions in order to meet the Process to Further Reduce Pathogens (PRFP) criteria prior to the windrow freezing in the winter. Since the climate conditions during this experiment were cold and wet and the moisture content was high in both windrows to start with, there was no need to add water, thus no noticeable difference in ability to insulate against evaporation or leaching of the slurry solids. The windrow with woodshavings as a bulking material did take longer to mature, but both composts produced were ready for application in the fall. Cumulative N gaseous losses were higher in the windrow with woodshavings with a total of 1.20 g N Mg⁻¹ (initial dry weight basis) (N₂O 1.19 g N Mg⁻¹ and NO 0.01 g N Mg⁻¹, NH₃ was not measured), compared to the windrow with straw, 0.73 g N Mg⁻¹ (N₂O 0.47 g N Mg⁻¹, NO 0.07 g N Mg⁻¹, and NH₃ 0.19 g Mg⁻¹). This was largely due to the production of N₂O from denitrification

when the windrow was turned and the center was anaerobic. Similarly, cumulative emissions of greenhouse gases, expressed as CO₂ equivalents were higher for the windrow with woodshavings, 1,126 kg Mg⁻¹ compared to 526 kg Mg⁻¹ for the windrow with straw as a bulking material; anaerobic conditions contributed to higher amounts of both CH₄ and N₂O. The compost produced from both types of bulking material met criteria for environmental safety as well as had desirable agronomic parameters for application or market. Due to the increased greenhouse gas and nitrogen gaseous losses, it was recommended to avoid using woodshavings as the only bulking material if other carbon sources are readily available.

4.2 Introduction

As discussed in Chapter 3, in Manitoba the most common practice for managing liquid pig slurry is application to agricultural soils nearby the pig production facility. However, this has led to a build-up up of phosphorus (P) in the soil and has been contributing to excess P runoff into waterways causing eutrophication (Statistics Canada, 2006). In order to minimize excess P, manure application regulations that were implemented in 2013 require manure application rates to be based on soil P levels instead of soil N levels. Essentially this would limit the amount of manure that can be applied each year requiring the pig producer to seek alternative management methods for manure. Centrifugation of the liquid slurry has been proposed as an option for pig producers who do not have the land base to comply with the regulations. Centrifuging liquid slurry separates the solid and liquid fractions. The solid fraction contains a higher portion of the P and has reduced volume and weight in comparison to the total, thus less costly to transport and can be managed separately. The liquid fraction, containing most of the water and a balanced concentration of nitrogen (N) and P, can continue to be applied to soils nearby. Chapter 3 verified that the solid

fraction, known as separated pig slurry solids, is a viable feedstock for windrow composting on-farm and can produce a marketable compost product. This value added compost product can be stored and shipped without odours to areas in Manitoba that require P. In Chapter 3, it was discussed that in order to stabilize N during the composting process the slurry solids require mixing with a bulking material to reduce the moisture content and provide carbon and structural support to maintain porosity; straw was used during that experiment. Straw was an adequate bulking material, it allowed completion of the composting process and produced compost that met maturity requirements based on temperature, carbon to nitrogen ratio (C:N), and extractable ammonium-nitrogen to nitrate/nitrite-nitrogen ratio ($\text{NH}_4\text{-N}:\text{NO}_3^-/\text{NO}_2^-\text{-N}$) by the spring of the following year. Despite meeting these indicators of maturity, at the end of the composting process straw fibres were still visible; generally, specifications for finished compost indicate there must be no identifiable material (Paul and Geesing, 2009). The concern with applying immature compost to an agricultural application is because microorganisms in the soil will use available nitrogen in the soil to further decompose the carbon, therefore depriving plants from utilizing soil nitrogen for crop production (Larney et al., 2006). It was unlikely that these small pieces of straw would have a major impact on nitrogen immobilization in soil, but screening would be required in order to sell the product for aesthetic reasons.

In Chapter 3 it was also discussed that the partial decomposition of straw could have been due to early loss of N therefore slowing the decomposition process. Early loss of N was attributed to poor contact between the straw and slurry solids leading to N losses through leaching of slurry solids and ammonia volatilization. Straw provided large pore spaces at the start of the composting process, these large pore spaces as well as heat from the composting process and weather conditions that were dry, hot, and windy rapidly evaporated moisture from the material required

the addition of moisture. When water was added the slurry solids, containing the majority of total N, washed off the straw due to the waxy texture. The large pore spaces also allowed ammonia volatilization to occur at the beginning of the composting process. The C:N at the start of the composting process was within the optimal range, but the carbon from the straw may not have been available to stabilize the N from the slurry solids if there was not good contact between the two feedstock materials. Another possibility that would cause only partial decomposition of the straw was small pile size and poor insulation of the microbial population in the winter, not allowing for recovery in the spring. There was an abrupt decline in compost temperature when the windrow was turned in the winter once mesothermic temperatures were reached. The windrows froze up and may have delayed re-establishment of the microbial population and further decomposition in the spring. The duration of the composting process could be extended to the fall to determine if more time would reduce the appearance of straw pieces, it would also allow observation throughout the summer to monitor the temperature of the windrow to determine if it is indeed fully matured.

The presence of straw fibre in the final compost was an indicator that straw may not be the most suitable bulking material for composting slurry solids. It was ideal to find a bulking material that produces a marketable value-added compost product but also limits both greenhouse gas and nitrogen gaseous losses during the composting process. Wood has been commonly used as a bulking material because it can be a natural biofilter for odours and can trap ammonia, waste wood is fairly available from tree trimmings or clean wood from construction or demolition projects (Environment Canada, 2013). Wood waste has been commonly used as a bulking material to compost municipal biosolids also known as sewage sludge. Biosolids are a similar feedstock to slurry solids because of the small uniform particle size, high moisture content, and high nitrogen

content. Typically, when biosolids are composted aeration technology has been utilized rather than turned windrow. This allows better control over odour and ammonia volatilization (Environment Canada, 2013). However, Nolan et al. (2011) and Huang et al. (2004) have successfully demonstrated the ability to compost pig slurry solids with sawdust using small tumblers or turned piles. Other benefits of using wood as a bulking material was the high C:N compared to straw, this reduces the volume of bulking material required which can reduce handling costs of transporting it to the site and turning the windrow, as well as the required area for composting (Environment Canada, 2013). A disadvantage of using woodshavings as a bulking material, compared to straw, may be a longer processing time. Woodshavings have a higher C:N, but the carbon comes in the form of lignin, which can be less available and takes longer to decompose compared to the carbon in straw, which was cellulose (Environment Canada, 2013; Paul and Geesing, 2009; Nolan et al., 2011; Hao et al., 2004). Typically, when wood was used as a bulking material with biosolids it was shredded to a mixture of ¼ inch to 3-inch particle sizes (Environment Canada, 2013). This provides heterogeneous material sizes to form pore spaces to allow controlled aeration. These pore spaces are maintained throughout the composting process, because the wood decomposes slowly; however, prior to application of the compost the large wood pieces are screened out to remove the excess carbon. Screening the material adds an additional step to the process, increasing the time and equipment costs. Instead, the particle size can be reduced by using woodshavings instead of woodchips to increase the availability of the carbon in wood by increasing the surface area to volume ratio (Nolan et al., 2011). This may allow the material to mature at a similar rate as if straw were used as a bulking material and avoid having to screen the finished compost prior to application. Other potential benefits of a smaller particle size and less porous material may include the ability to control fluctuations against the weather, preventing evaporation of water when

weather is hot, dry, and windy as well as insulate the heat within the windrow during winter to keep microorganisms active longer into the winter. The small particle size of woodshavings are also abrasive and could provide better contact with slurry solids compared to straw, if water needs to be added the slurry solids may not be leached out. However, using woodshavings does not provide the structure for pore spaces and there could be a higher potential for anaerobic conditions generating methane and potentially N losses through denitrification related nitrous oxide (N₂O) emissions, especially using turned windrow composting technology.

In Chapter 3, the gaseous emissions that were measured were compared to Total C and Total N losses calculated by mass balance and were found to be underestimated. The mass losses in Chapter 3 were significant and could be reduced by removing less material when sampling and turning the windrow. It was also suggested that the gaseous emissions were underestimated because the experiment was conducted in an open area and wind effected the ability for the chambers to capture the gaseous fluxes. Therefore, this experiment will be conducted in a sheltered area and gaseous emissions measured will also be compared to Total C and Total N losses calculated by mass balance to determine if this improves the capability of the automated chamber and FTIR system to capture gaseous emissions.

In order to sell the finished compost product additional environmental safety parameters must also be considered to comply with Canadian Food Inspection Agency (CFIA), which includes meeting requirements of the Canadian Council of Ministers of the Environment (CCME) Compost Quality Guidelines. Parameters to be considered for the final compost product include compost maturity, the presence of foreign materials, pathogens, and trace elements. To meet CCME Compost Quality Guideline maturity requirements carbon dioxide (CO₂) respiration can be conducted on the final compost product. In addition, maturity can be defined by a Solvita test,

which gives a indication of microbial activity and ammonia volatilization, as well as C:N, $\text{NH}_4\text{-N:NO}_3^-/\text{NO}_2^-\text{-N}$, and temperature to be consistent with information from Chapter 3. To meet the CCME Guidelines for Compost Quality pathogen requirements the temperature and number of turns must be recorded to determine if the Process to Further Reduce Pathogens (PFRP) has been met, the final compost product must also be tested for levels of Escherichia coli, fecal coliform and salmonella. Other safety concerns for the final compost product include testing for the presence of foreign material and trace elements including cadmium, cobalt, chromium, copper, mercury, molybdenum, nickel, lead, selenium, zinc, arsenic, mercury, selenium. In order to market the product and compare the nutrient value, potential for salt toxicity, and ease of use the producer should know certain agronomic parameters as well, including total N, total P, total potassium (K), organic matter (OM), pH, electrical conductivity (EC), total sodium (Na), % base saturation Na, bulk density (BD), MC, and particle size Providing consumers with this information ensures confident purchasing of the product.

Therefore, the objectives of this study were to:

- 1) Determine the suitability of woodshavings compared to straw as a bulking material to compost separated pig slurry solids in order to provide better insulation throughout the winter to produce mature compost ready for use in the fall.
- 2) Determine the effect of woodshavings compared to straw as a bulking material on greenhouse gas and nitrogen gaseous losses when used to compost separated pig slurry solids. As well as compare total C and N emissions to total C and N calculated by mass balance to evaluate the capability of the automated chamber and FTIR system in a sheltered area.

- 3) Determine the effect of woodshavings compared to straw as a bulking material on overall quality of composted pig slurry solids in terms of compost maturity indicators (CO_2 respiration, Solvita, C:N, $\text{NH}_4\text{-N}:\text{NO}_3\text{-N}$, temperature), environmental safety parameters (trace elements, presence of foreign materials, pathogens), and agronomic parameters (Total N, Total P, Total K, OM, pH, EC, Total Na and % base saturation Na, BD, MC, and particle size).

4.3 Materials and Methods

4.3.1 Feedstock Sources

The pig slurry was from a conventional farrow to finish pig barn, different from the source in Chapter 3, near Niverville, Manitoba with a pull-pit collection system that drained into a wet well. Slurry solids were obtained by centrifugation of pig slurry without the addition of any polymers or flocculants using a two-phase clarifying decanter (ALSYS 20, Alfa Laval Inc. Scarborough, ON). The solids were stockpiled over a period of only one week before use in one of the two windrows. The amount of slurry solids available was limited because the decanter was recently moved to the pig barn and there were delays in producing the slurry solids prior to starting the composting trial.

Wheat straw was added to one of the windrows as a bulking material. The wheat straw was from one-year old bales from the Glenlea Research Station near Glenlea, Manitoba, the bales were wetted prior to use. The other windrow used woodshavings as a bulking material. The woodshavings were a mix of 0.2 to 2 cm pieces. The woodshavings were from Floodway Animal

Bedding Ltd., in Selkirk, Manitoba. They were freshly chipped and a mixture of soft and hard woods including birch, pine, oak, aspen, and cedar, the ratio of each was not known.

Samples of the feedstock materials, straw, woodshavings, and slurry solids, were taken as each material was added to the feedmixer, approximately 30 samples 1 L in size. Samples were mixed together to form a composite sample then a 2 L sub-sample was taken by mixing and quartering down the composite sample. Re-mixing the composite sample and repeating the quartering process obtained a duplicate sample. The two samples were frozen prior to shipping to A&L Laboratories (London, ON) and analyzed for BD, MC, C:N, and Total N, Total phosphorus P, and Total K concentrations. The woodshavings had a higher BD, higher C:N, lower MC, and supplied lower Total N, Total P, and Total K compared to straw (Table 4.1). In comparison to Chapter 3, the slurry solids used for this experiment had a higher BD, a higher C:N, higher MC, and lower Total N; Total P and K were similar (Table 3.1 and 4.1).

Table 4.1 Characteristics of feedstock materials including straw, woodshavings and slurry solids.

Material	BD kg m ⁻³	C:N Ratio	MC	Total N g kg ⁻¹	Total P	Total K
Straw	88 (7)	61 (8)	487 (91)	8.0 (0.9)	1.0 (0.1)	15.7 (1.7)
Woodshavings	117 (2)	533 (120)	45 (1)	1.1 (0.2)	BDL	1.6 (0.2)
Slurry Solids	399 (22)	20 (1)	690 (8)	23.5 (0.9)	20.4 (0.1)	6.3 (0)

Values are the average of two sub-samples ± 1 standard error in parenthesis. Values for bulk density (BD) and moisture content (MC) are reported on an as-is basis and those for Total nitrogen (N), phosphorus (P), and potassium (K) on a dry weight basis.

4.3.2 Compost Setup and Windrow Management

The two windrows were initiated on October 15, 2012 (DOY 289). A feedmixer (900T, Supreme Int., Wetaskiwin, AB) was used to mix the slurry solids with either the straw or woodshavings.

The windrows were setup on a farmyard near La Broquerie, Manitoba. To minimize the loss of gas flux capture, through horizontal wind movement, the location of the windrows was sheltered from the wind with a barn to the northwest and a tree line along the east and south, and the windrows were aligned in a west-east direction to show the least surface area to the non-sheltered side.

In total, the window with straw as a bulking material had 816 kg of straw and 622 kg of slurry solids. The windrow with woodshavings as a bulking material had 520 kg of woodshavings and 748 kg of slurry solids, on a dry weight basis. The mixing ratio of straw or woodshavings to slurry solids was 1.3 and 0.69 respectively (Table 4.2). The starting MC and C:N for these windrows started higher than the ideal range, but they were comparable to each other and there was little variability between samples (Table 4.2).

Table 4.2 Added amount of feedstock materials, straw or woodshaving bulking material and slurry solids, and characteristics of starting mixture on October 15, 2012 (DOY 289).

Treatment	Added Feedstock Material		Starting Mixture		
	Bulking Material	Slurry Solids	BD kg m ⁻³	MC g kg ⁻¹	C:N Ratio
Straw	567	433	151 (21)	642 (6)	32 (2)
Woodshavings	410	590	346 (4)	613 (10)	36 (0)

Values are the mean of three samples ± 1 standard error in parenthesis. Values for added feedstock material is on a dry weight basis and those for Bulk Density (BD) and Moisture Content (MC) are reported on an as-is basis.

The windrow containing straw was longer than the windrow containing wood shavings because more straw was needed to reach the desired C:N, and straw has a lower bulk density compared to wood shavings. The length of the windrow with straw and woodshavings as a bulking material was 12.5 m and 4.9 m, respectively. The height and width of the windrows were the same

at 1.2 m and 2.4 m respectively. The windrows were turned by a composting service co-operative, Compo-stages, with a Backhus windrow compost turner (17.50, Backhus EcoEngineers, Eggersmann Group, Edeweicht, Germany). The Backhus turner goes through the compost windrow leaving the ground underneath undisturbed. Windrow turning occurred on October 22 (DOY 296), October 30 (DOY 304), November 6 (DOY 311), November 15 (DOY 320), January 8 (DOY 374), May 9 (DOY 495), June 14 (DOY 531), August 16 (DOY 594), and September 6 (DOY 615). Therefore, the windrows were turned nine times. Addition of water was not needed throughout the experiment however, rain and snow was incorporated into the windrows increasing the moisture content. Throughout the experiment, October 15, 2012 (DOY 289) to September 6, 2013 (DOY 615), the area received 297 mm of total precipitation (weather station located at the Trace Gas Manitoba Research Site, Glenlea MB, approximately 60 kilometers west of the composting site); which was less than the 30-year average (1981-2010) Climate Normal of 521 mm, measured at the Winnipeg Richardson International Airport, Winnipeg Manitoba (Environment Canada, 2014).



Figure 4.1 Turning the windrows with a Backhus windrow turner on October 22, 2011 (DOY 296).

On January 8 (DOY 374) a front-end loader was used to turn the windrows in an attempt to pile them up larger to provide better insulation. However, this was not an effective turning method because the outer surface on the compost windrow was frozen resulting in large clods instead of mixing. After that turning event the windrows were covered until spring with a geotextile polypropylene and polyester fabric (Compostex, Texel, Saint-Elzear, QC) that allows the compost to exchange air but protects against water.

4.3.3 Compost Sampling and Analysis

Samples for analysis at the Soil Ecology Laboratory (Soil Science Department, University of Manitoba, MB) were taken immediately after a windrow was turned as well as an additional 1-2 times per week from October 15 (DOY 289) to January 10 (DOY 374). Starting May 9 (DOY 495)

samples were taken only immediately after turning the windrow until September 6 (DOY 615). Samples were analyzed for MC, EC, pH, $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-/\text{NO}_2^-\text{-N}$. Methodology for analyzing the samples is described in Chapter 3, the Test Methods for the Evaluation of Composting and Compost (TMECC) were used (Thompson et al., 2001).

Samples for analysis at A&L Laboratories were taken immediately after each time the windrow was turned from October 15 (DOY 289) to August 16 (DOY 594). Samples were analyzed for BD, C:N, organic carbon, and total N, P, K, and Na. Methodology for analyzing the samples is described in Chapter 3, the Test Methods for the Evaluation of Composting and Compost (TMECC) were used (Thompson et al., 2001).

Samples were obtained by taking twenty random grab samples across the windrow approximately 10-60 cm below the surface. The grab samples were mixed and quartered down to make three 2 L sub-samples and three 200 g sub-samples. The unused material was returned to the respective windrow. All samples were frozen at $-18\text{ }^\circ\text{C}$ prior to analysis. The three 200 g samples from the windrow with straw as a bulking material, that were analyzed by the Soil Ecology Laboratory, had the straw pieces cut using scissors, until almost all of the straw was 1cm or less in length, prior to analysis.

For each of the sampling dates, the three samples were treated as replicates, the average and standard error were determined for each parameter. The final compost samples were taken on August 16, 2013 (DOY 594). Statistical analyses were performed on the results of the final compost samples with SigmaPlot Systat Software (SigmaPlot for Windows Version 11.0, Germany). A two-way t-test ($P<0.05$) was used to determine significant differences due to the effect of straw or woodshavings as a bulking material for each of the parameters. Data was tested

for normality (Shapiro-Wilks) and equal variance. Data for pH failed the normality test; therefore, a Mann-Whitney Rank Sum Test was used.

There was also one representative final compost sample of each windrow taken on September 6 (DOY 615) and sent to A&L labs for the “Compost Quality Alliance” test package. This tested for environmental safety parameters as described in the CCME Compost Quality Guidelines including maturity, presence of foreign material, trace elements, and pathogens as well as agronomic parameters including particle size, cation exchange capacity, and percent base saturation of Na, K, magnesium (Mg), and calcium (Ca). Additionally, as part of the Compost Quality Alliance package A&L labs gives an overall recommendation for suitable uses for the material based on the overall quality of the final product. Test method for maturity included analysis of CO₂ evolution rate respiration-CO₂-C g⁻¹ OM day⁻¹ (TMECC 5.08-B; Thompson et al., 2001) reported in mg CO₂ as well as Solvita Maturity Index (TMECC 5.08-E; Thompson et al., 2001). The presence of foreign material including total, plastics, and sharps was analyzed using TMECC 3.08 (Thompson et al., 2001). A 500 ml sample was analyzed on an as-is basis; the number of foreign materials was counted as well as the weight of material in % reported on a dry weight basis. Analysis for trace elements required air-dried milled samples to be digested with nitric acid (TMECC 4.12-B nitric acid digestion protocol; Thompson et al., 2001). Then the concentration in ug g⁻¹ of cadmium, cobalt, chromium, copper, mercury, molybdenum, nickel, lead, selenium, and zinc was determined by inductively coupled plasma (ICP) spectrometry (TMECC 4.14; Thompson et al., 2001). The concentration in ug g⁻¹ of arsenic, mercury, and selenium was determined by atomic adsorption spectrometry (TMECC 4.13-B; Thompson et al., 2001). Analysis of pathogens including *Escherichia coli* and fecal coliform were determined using the Health Protection Branch test protocols Microbiology Food Health Protection Branch

(MFHPB)-19 and reported in units of MPN g⁻¹ on a dry weight basis. Salmonella spp. was determined using the Health Protection Branch MFLP-75 protocol and reported in units of presence-absence (P-A). Particle size was determined by American Society for Testing and Materials (ASTM) D422, the screens used were 2, 1, ½, 3/8, and ¼ inch in size. A&L labs would not provide details for extracting the soluble Na, K, Mg, and Ca in order to calculate total cation exchange capacity and % base saturation.

4.3.4 Compost Temperature, Air Temperature, Oxygen Concentration and Wind Monitoring

The temperature of the compost windrow was monitored every half-hour from October 17 (DOY 291) to September 6 (DOY 615) using thermocouples attached to a CR1000 data logger and multiplexer (Campbell Scientific Canada, Edmonton, AB). Thermocouple dowels were inserted perpendicular to the side of windrow; measurements were taken 5, 25, 45, 65 cm from core to edge of window. There were four thermocouple dowels per windrow. Temperature was determined by averaging the temperatures of the four dowels at the four depths. To determine if the PFRP was met, the depth with the highest average temperature from the four dowels was used. To successfully meet PFRP criteria, compost temperature must be above 55 °C for 15 days and turned 5 times within that period.

Air temperature was measured every half-hour from October 17 (DOY 291) to September 6 (DOY 615) using a Type T thermocouple mounted 0.6 m above the ground, sheltered from sunlight and wind. The 30-year average (1981-2010) Climate Normal air temperature was measured at the Winnipeg Richardson International Airport, Winnipeg Manitoba (Environment Canada, 2014).

Measurements of % oxygen were taken from the center of each windrow at three locations using a Model OT-21 oxygen monitor (Geneq Inc., Montreal, QC) one to three times per week from October 24 (DOY 298) to August 26 (DOY 604).

A wind monitor (05103-10, R.M. Young Company, Traverse City, MI, USA) was set up next to the windrows 1.5 m off the ground, to measure wind speed every half hour from October 18 (DOY 292) to September 6 (DOY 615).

4.3.5 Gas Flux Measurement

An automated gas flux chamber system (Long-term Automated Chambers, LICOR BioSciences Inc., Lincoln, NB, USA) and a Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer (DX4015, Gaset Technologies Inc., Helsinki, FIN) was used to determine gas fluxes from the windrows. As described in Chapter 2 and 3, the two components were placed in series to create a closed loop system that collected and measured the accumulation of gas emitted from the windrow when one of the eight chamber lids were deployed. The FTIR was capable of simultaneously measuring CO₂, CH₄, N₂O, NO₂, NO, and NH₃ volumetric concentrations.

The chambers were set up on the top of the windrows, initially four chambers were set up on each of the windrows on October 17 (DOY 291); on November 15 (DOY 320) the windrow with woodshavings as a bulking material had decreased in length and only two chambers would fit on the top of the windrow. On May 9 (DOY 495) the windrow with straw as a bulking material had decreased in size to only allow two chambers to fit. Only one chamber observation could occur at a time. The sequence was set up to alternate measurements between windrows. The chamber deployment time started at one minute with a purge time of twenty-nine minutes but increased to a three-minute deployment time and twenty-seven-minute purge time after November 1 (DOY

306), therefore a measurement was taken every thirty minutes. Gas fluxes were monitored from October 17 (DOY 291) to August 26 (DOY 604) for a total of 313 days.

4.3.6 Gas Flux Calculation and Cumulative Emission Estimation

In Chapter 2, the gas flux calculation procedure is given in detail. In brief, the Ideal Gas Law ($PV=nRT$) was used to convert concentration (ppm) of gas in the total flux system volume into mass of carbon (g C) or nitrogen (g N). Then gas fluxes were estimated in g C or N $m^{-2}s^{-1}$ from the linear change in gas concentration over time and the surface area of the chamber collar.

Each chamber was a replicate unit; daily gas flux was calculated by averaging all fluxes for a chamber over the course of a day. The average and one standard error of the average were determined for each day from the daily gas flux for individual chambers.

Adding the daily gas flux for each chamber and then averaging those values calculated cumulative emissions for each windrow in g C or N m^{-2} . When a chamber was taken out of the Trial the cumulative sum was averaged at that point and that value was used to continue for the next interval. Linear interpolation of gaps in the daily gas flux data sets was used for periods when monitoring did not occur, such as of a power outage on December 14 (DOY 349) to December 16 (DOY 351), as well as when the FTIR analyzer was being used for alternative applications in the spring (45 non-consecutive days from May 14 (DOY 500) to August 22 (DOY 600)). Gas fluxes were not measured from January 8 (DOY 374) to May 9 (DOY 495), because the windrows were covered with the fabric material during the winter. It was assumed that there were zero gas fluxes during this time based on the compost temperatures and the relatively low gas fluxes prior to and after this time period.

The mass of gases emitted (g-C or g-N) was calculated by multiplying the surface area of the windrow by the daily average gas flux and then adding these values for each day during the experiment duration. The total surface area of the windrows was calculated after each time the windrow was turned. Surface area was calculated using the formula for compost windrows in between an oval and a trapezoid shape as described in Paul and Geesing (2009). Total mass of gases emitted was also expressed based on the initial dry mass of the windrow, kg C or N Mg⁻¹. The sum of g-C from CO₂ and CH₄ emissions was used to calculate total g-C of gases emitted, and then the percentage of each contributing to the total was determined. Similarly, the sum of g-N from N₂O, NO, and NH₃ emissions was used to calculate total g-N of gases emitted; then the percentage of each contributing to the total was determined.

Statistical analysis could not be conducted to compare the gas flux emissions from the two windrows. The amount of pig slurry solids available at the time of the experiment limited the size of the windrows and on November 15 (DOY 320) only two chambers would fit on the top of the windrow with woodshavings. Due to the low number of chambers it was not appropriate to treat the chambers as pseudo replicates.

4.3.7 Estimated Losses of Mass, Carbon and Nitrogen Based on Mass Balance

As discussed in Chapter 3, Total C and Total N loss based on mass balance was estimated by subtracting the Total C or Total N within the final material from the Total C or N within the starting material. Total C was calculated by multiplying the initial or final windrow mass, on a dry mass basis, by the initial or final value of organic carbon determined in the samples sent for analysis. Total N was calculated similarly by multiplying the initial or final windrow mass, on a dry mass basis, by the initial or final value of Total N determined in the samples sent for analysis. The final

mass, on a dry mass basis, was calculated based on estimations of volume, calculated using the formula for surface area of a windrow with a shape in between an oval and a trapezoid (Paul and Geesing, 2009), and bulk density and moisture content analysis. The values for mass, total C and total N were expressed as the amount per initial windrow dry mass (g Mg^{-1}). The loss of mass, loss of Total C and loss of Total N was also expressed as a percentage of the initial dry mass, initial Total C, or initial Total N, respectively. The total mass of gaseous emissions measured were also expressed as a percentage of initial Total C and initial Total N, as well as the percentage of Total C or N loss.

4.4 Results

4.4.1 Air Temperature and Precipitation During the Experiment

From October 2012 to September 2013 the air temperature was normal compared to the 30-year (1981-2010) Canadian Climate Normals for the Winnipeg Richardson International Airport, located approximately 75 kilometers north west of the composting site (Environment Canada, 2014). Average daily air temperature was around 0 °C throughout October and steadily declined to -15 °C in January, with a few daytime lows reaching -35 °C (Figure 4.1). Average daily temperature started to increase after mid March continuing throughout the summer.

Precipitation in October was slightly higher than the 30-year (1981-2010) Canadian Climate Normals for the Winnipeg Richardson International Airport (Environment Canada, 2014). Very little precipitation was measured for the months of December 2012 and January 2013; this value may be underestimated due to equipment failure or rounding errors, because snow was observed frequently and considerable amounts during this time. During the remainder of the

experiment period precipitation was slightly below but followed the trend of the 30-year (1981-2010) Canadian Climate Normals for the Winnipeg Richardson International Airport (Environment Canada, 2014).

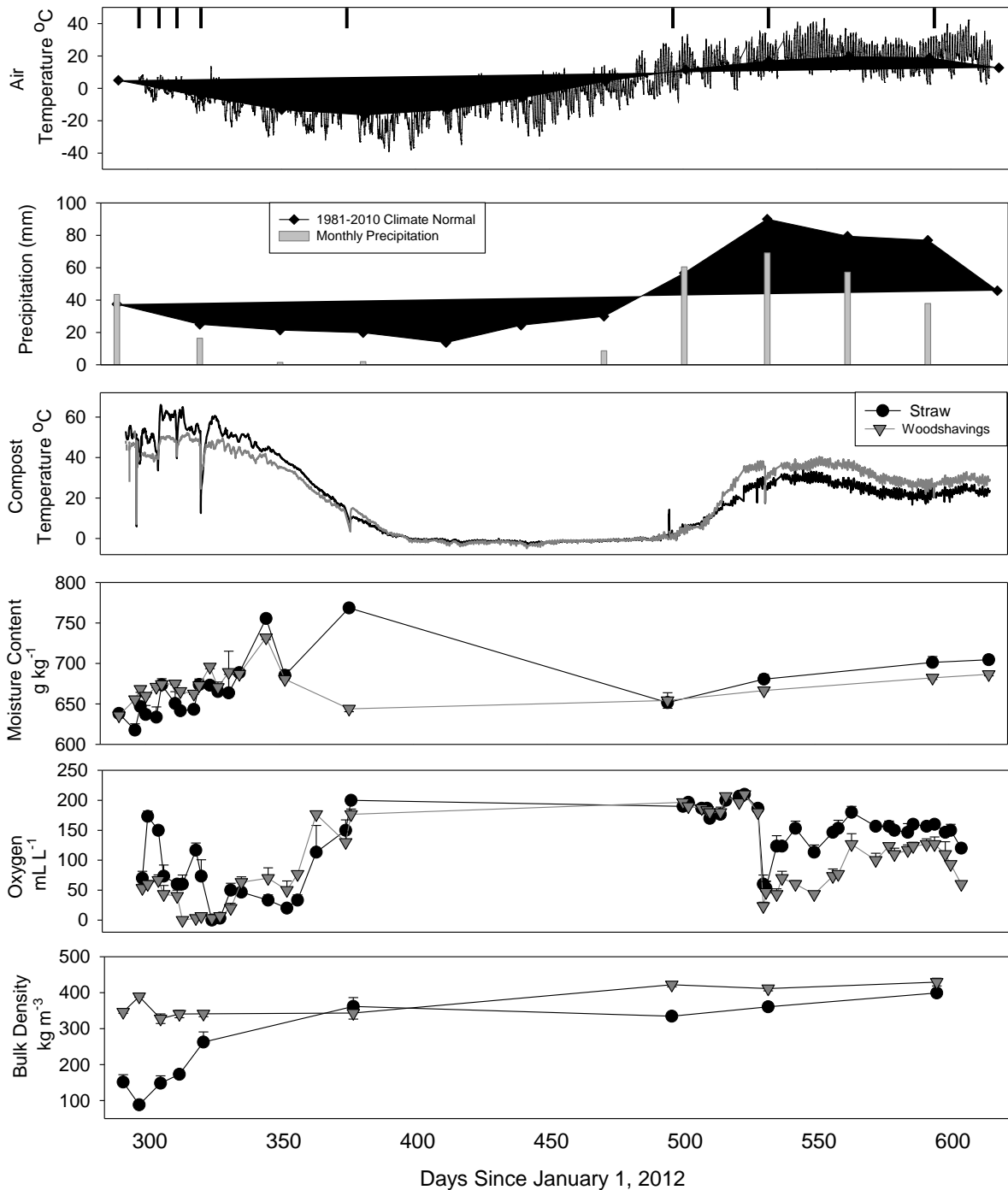


Figure 4.2 Measured air temperature and monthly precipitation and 30-year (1981-2010) Winnipeg Richardson International Airport Climate Normals for average air and precipitation, compost windrow temperature, moisture content (MC), oxygen concentration, bulk density (BD), and electrical conductivity (EC) for windrows with straw and woodshavings as bulking material from October 16, 2012 (DOY 290) to September 6, 2013 (DOY 615). Values for air temperature are based on one measurement whereas compost temperature is the mean of all measurements throughout the windrow. Values for MC, oxygen concentration, BD, and EC are the mean of three samples and bars where shown are ± 1 standard error. MC and BD are reported on an as-is basis. Bars indicate windrow turning.

4.4.2 Compost Properties During the Experiment

4.4.2.1 Compost Temperature

The entire thermophilic stage, in which average temperatures were above 50 °C, for the windrow with straw as a bulking material occurred on 45 days between October 17 (DOY 291) to December 12 (DOY 347), total duration of 57 days, in which the windrow was turned 4 times (Figure 4.1). The thermophilic stage for the windrow with woodshavings as a bulking material occurred on 13 days in-between October 18 (DOY 292) to November 11 (DOY 316), for a total duration of 25 days, in which the windrow was turned three times. Using only the highest recorded temperature to determine if PFRP requirements were met, the windrow with straw as a bulking material was above 55 °C for 60 consecutive days in-between October 17 (DOY 291) to December 15 (DOY 350), and was turned four times during this period. The windrow with wood as a bulking material was above 55 °C for 44 non-consecutive days, including October 18 (DOY 292) to October 30 (DOY 304), November 2 (DOY 307) to November 15 (DOY 320), and November 20 (DOY 325) to December 6 (DOY 341), in which it was turned 4 times.

The two windrows exhibited a similar temperature pattern throughout the composting process. At the start of the composting process thermophilic temperatures occurred right away. The windrow with woodshavings as a bulking material was 5 to 10 °C below the average temperature of the windrow with straw bulking material and was also slower to reach thermophilic temperatures after turning. During the thermophilic stage in the windrow with woodshavings as a bulking material it was observed that the temperature probe at 45 cm from the surface was higher than the temperature probe at 65 cm (the core of the windrow) from the surface. In the windrow with straw as a bulking material the temperature probes showed the core of the windrow to have the highest temperature and slightly cooler towards the edge of the windrow during the

thermophilic phase. During the thermophilic phase the compost temperature was not affected by daily air temperature fluctuations. Both windrows started to slowly decline in temperature after December 2 (DOY 328), but remained in mesothermic temperatures. After the windrows were turned on January 8 (DOY 374) they did not heat up, instead the temperature continued to drop until the piles were below freezing temperatures throughout the winter. On May 13 (DOY 499), four days after turning the windrows in the spring the compost temperature started to increase to ambient air temperature. The temperature of the windrow with straw as a bulking material stayed around the same temperature as ambient, temperatures at the core of the windrow remained steady while the outer edge of the windrow fluctuated with day and night time air temperatures. The temperature of the windrow with woodshavings as a bulking material was higher than the temperature of the windrow with straw, up to 15 °C higher from May 27 (DOY 513) to August 1 (DOY 579); by August 16 (DOY 595) the windrow temperature was only 5 °C higher. The internal temperatures were higher and the outer edge temperature of the windrow with wood shavings as a bulking material did not fluctuate as much as the windrow with straw as a bulking material.

4.4.2.2 Compost Moisture Content

Moisture content for both of the windrows started at 642 g kg⁻¹ and 613 g kg⁻¹ for the windrow with straw and woodshavings as a bulking material, respectively (Figure 4.1), these values are above the range of 500 to 600 g kg⁻¹ (as-is basis), which is generally considered optimal for composting during the active stage (Environment Canada, 2013). Moisture content continued to increase as precipitation occurred and snow was incorporated into the windrow. The maximum moisture content was 769 g kg⁻¹ and 732 g kg⁻¹ on January 10 (DOY 376) and December 10 (DOY 345) for windrows with straw and woodshavings as bulking materials, respectively. Moisture

content did decrease but remained above the optimal range even during the curing stage of the composting process, despite the only addition of water being precipitation. The final moisture content on September 6 (DOY 615) was 705 g kg⁻¹ and 687 g kg⁻¹, for windrows with straw and woodshavings as bulking materials, respectively.

4.4.2.3 Compost Oxygen Concentration

The oxygen concentration for the windrow with straw as a bulking material was limited, below 130 mL L⁻¹, during the thermophilic stage of the composting process. However, turning the windrow had an effect and showed replenished oxygen immediately afterwards (Figure 4.1). The oxygen content increased after December 21 (DOY 356) to near ambient levels and remained at that level into the spring until after turning the windrow on June 13 (DOY 530). Thereafter, oxygen concentration was slightly below ambient, but remained above 130 mL L⁻¹ to the end of the experiment.

The oxygen concentration for the windrow with woodshavings as a bulking material the had oxygen content well below 130 mL L⁻¹ during the thermophilic stage and turning did not replenish oxygen (Figure 4.1). The oxygen concentration began to increase to 177 mL L⁻¹ on December 21 (DOY 356) and remained at that level until spring when the windrow was turned on June 13 (DOY 530). Oxygen concentration was limited thereafter remaining below 130 mL L⁻¹ to the end of the experiment.

On January 10 (DOY 376) the outer shell of the windrows were covered with fabric material and the oxygen probe was not inserted, therefore measurements were not taken after that date until May 14 (DOY 500).

4.4.2.4 Compost Bulk Density

Bulk density (BD) for the windrow with straw as a bulking material started at 151 kg m^{-3} and ended at 399 kg m^{-3} on August 16 (DOY 594). There was a steady increase in BD until January 10 (DOY 376); 87 days after initiating, and then remained at a steady state (Figure 4.1). BD for the windrow with woodshavings as a bulking material started at 346 kg m^{-3} and gradually increased to 429 kg m^{-3} (Figure 4.1).

4.4.2.5 Compost pH

The pH starts at 8.03 and 7.93 for the windrows with straw and woodshavings as bulking materials, respectively. There is a similar trend to the pH values for both the windrows. There is a slight decrease after the first turning event on October 22 (DOY 296) as well as after the fourth turning event on November 15 (DOY 320). The final pH is 6.1 for both the windrows with straw and woodshavings as bulking material.

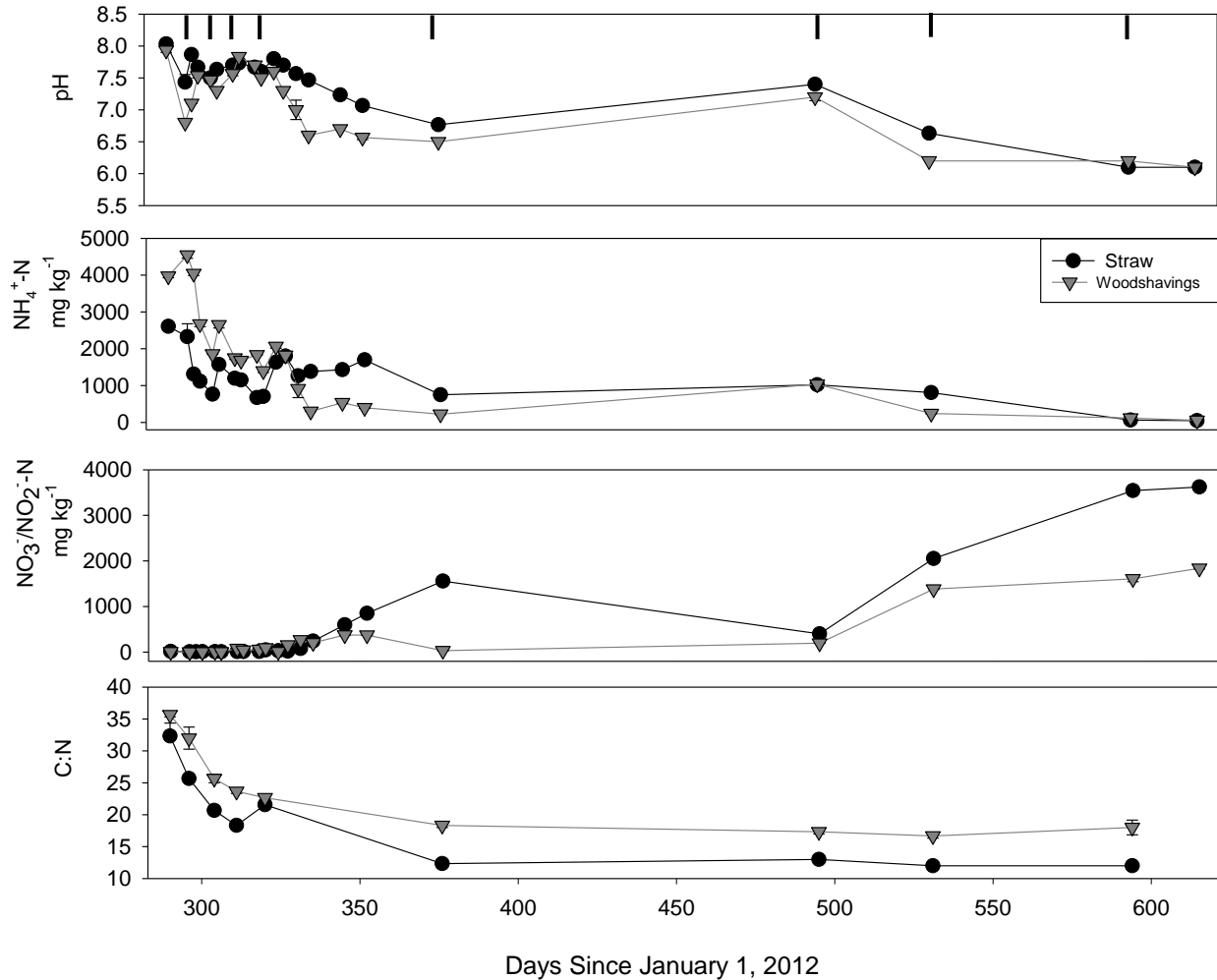


Figure 4.3 pH, ammonium ($\text{NH}_4^+\text{-N}$), nitrate/nitrite ($\text{NO}_3^-/\text{NO}_2^-\text{-N}$), and carbon to nitrogen ratio (C:N) for windrows with straw and woodshavings as bulking material from October 16, 2012 (DOY 290) to September 6, 2013 (DOY 615). Values for pH, $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-/\text{NO}_2^-\text{-N}$, and C:N are the mean of three samples and bars where shown are ± 1 standard error. $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-/\text{NO}_2^-\text{-N}$ are reported on an as-is basis. Bars indicate windrow turning.

4.4.2.6 Compost Ammonium and Nitrate/Nitrite

The concentration of $\text{NH}_4^+\text{-N}$ started at $2,607 \text{ mg kg}^{-1}$ and $3,973 \text{ mg kg}^{-1}$ for the windrows with straw and woodshavings as bulking materials, respectively (Figure 4.2). In both windrows $\text{NH}_4^+\text{-N}$ rapidly dropped by November 5 (DOY 310), and gradually continued to decrease until reaching 47 mg kg^{-1} and 56 mg kg^{-1} on September 6 (DOY 615) for windrows with straw and woodshavings as bulking material, respectively.

The concentration of $\text{NO}_3^-/\text{NO}_2^-$ -N started at 15.6 mg kg^{-1} and 8 mg kg^{-1} for the windrows with straw and woodshavings as bulking materials, respectively. In both windrows, $\text{NO}_3^-/\text{NO}_2^-$ -N started to increase on December 10 (DOY 345), especially in the straw pile, but came back down when analyzed at the start of spring. $\text{NO}_3^-/\text{NO}_2^-$ -N started to rapidly increase until the final analysis on September 6 (DOY 615) at $3,623 \text{ mg kg}^{-1}$ and $1,840 \text{ mg kg}^{-1}$ for windrows with straw and woodshavings as bulking material, respectively.

4.4.2.7 Compost Carbon to Nitrogen Ratio

The C:N for the windrows with straw and woodshavings as bulking material started slightly above the optimal range at 32.3 and 35.7, respectively (Figure 4.2). For both the windrows the C:N decreased rapidly until January 10 (DOY 376), 87 days after initiating the windrows, and then levelled off. The final C:N of the windrows with straw and woodshavings as a bulking material was 12 and 18, respectively.

4.4.3 Greenhouse Gas and Nitrogen Gaseous Losses

4.4.3.1 Daily Mean Gas Fluxes for the Windrow with Straw as a Bulking Material

Fluxes of CO_2 from the windrow with straw as a bulking material were high immediately after the composting process was initiated (Figure 4.3). After the first windrow turning, October 22 (DOY 296) until the second windrow turning, October 30 (DOY 304), CO_2 fluxes were lower and at a steady state with little variability. After turning the windrow on October 30 (DOY 304), there was an immediate response in increased CO_2 fluxes, as well as after the third and fourth turnings, November 6 (DOY 311) and November 15 (DOY 320). The highest daily flux peaked on November 18 (DOY 323) at $1,072 \text{ g C m}^{-2} \text{ d}^{-1}$. There was a gradual decline in CO_2 fluxes thereafter.

There was also an increase in CO₂ fluxes not initiated by turning that started to rise on December 6 (DOY 341), peaked on December 9 (DOY 344) at 576 g C m⁻² d⁻¹ and had significantly decreased to 10 g C m⁻² d⁻¹ by January 8 (DOY 374) when the windrow was last turned in the winter. After turning the windrow in the spring, CO₂ fluxes started below 10 g C m⁻² d⁻¹ but then started to increase on May 28 (DOY 514) reaching 80 g C m⁻² d⁻¹ by June 28 (DOY 545), but had returned to below 10 g C m⁻² d⁻¹ by August 26 (DOY 604). Therefore, the active stage of composting, considered as the span of time CO₂ fluxes were high and indicative of high microbial activity, was considered from October 17 (DOY 291) until December 25 (DOY 360) for a total of 69 days, ranged from 5 g C m⁻² d⁻¹ to 1,072 g C m⁻² d⁻¹ with an average of 348 g C m⁻² d⁻¹.

Fluxes of CH₄ did not occur until October 31 (DOY 305), one day after the second turning the windrow on October 30 (DOY 304), peaking on November 3 (DOY 308) at 24.66 g C m⁻² d⁻¹ and then declined by November 13 (DOY 318). Fluxes of CH₄ also occurred on November 16 (DOY 321), one day after the fourth turning of the windrow on November 15 (DOY 320), and peaked at 66.63 g C m⁻² d⁻¹ on November 19 (DOY 324), then decreased to baseline values by December 15 (DOY 350) and remained negligible for the remainder of the composting process.

Fluxes of N₂O started after the third window turning, beginning to increase on November 11 (DOY 315) and continued until January 8 (DOY 374), when the windrow was last turned in the winter. The highest daily N₂O flux during this time was 1.41 g N m⁻² d⁻¹ on December 9 (DOY 344). In the spring there was some slight N₂O fluxes from June 2 (DOY 519) to July 2 (DOY 549), but remained less than 0.38 g N m⁻² d⁻¹.

Fluxes of NO started later in the composting process on November 23 (DOY 328) and peaked on December 9 (DOY 344) at 0.42 g N m⁻² d⁻¹, declining thereafter but continued at a low

level above zero until the windrow was turned on January 8 (DOY 374). NO fluxes were negligible for the remainder of the composting process.

Fluxes of NH₃ started immediately during the composting process, peaking at 4.35 g N m⁻² d⁻¹ on the first day, October 17 (DOY 291), but had declined rapidly by October 21 (DOY 295) and remained negligible thereafter. Values of NO₂ were negligible or negative; there were no observable trends.

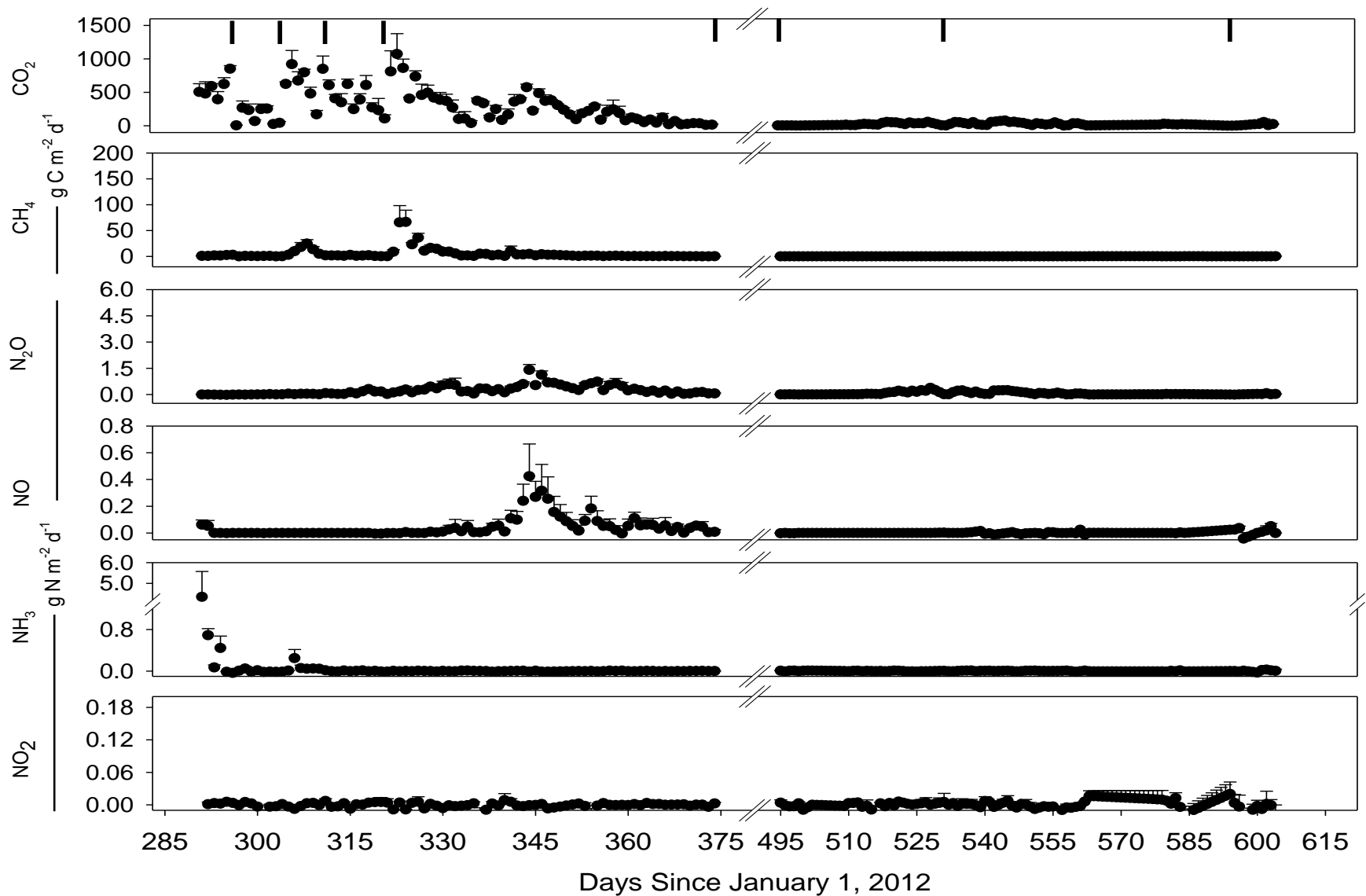


Figure 4.4 Daily mean gas fluxes and where shown ± 1 standard error for the windrow with straw as a bulking material from October 17, 2012 (DOY 291) to September 6, 2013 (DOY 615). Bars indicate windrow turning.

4.4.3.2 Daily Mean Gas Fluxes for the Windrow with Woodshavings as a Bulking Material

Fluxes of CO₂ from the windrow with woodshavings as a bulking material did occur immediately, but were lower relative to what was measured later in the composting process. Fluxes of CO₂ started to increase after the second turning of the windrow on October 30 (DOY 304), peaking at 953 g C m⁻² d⁻¹ on November 1 (DOY 306). Fluxes of CO₂ started to decline on November 11 (DOY 316) gradually down to 10 g C m⁻² d⁻¹ by January 8 (DOY 374), when the windrow was last turned in the winter. After turning the windrow in the spring on May 9 (DOY 495), CO₂ fluxes were below 10 g C m⁻² d⁻¹ but started to increase by May 17 (DOY 503) reaching 140 g C m⁻² d⁻¹ by June 4 (DOY 521), and declined to below 10 g C m⁻² d⁻¹ by August 26 (DOY 604). The active stage of composting, indicated by high CO₂ fluxes, ranged from October 17 (DOY 291) to December 16 (DOY 351), for a total of 60 days that ranged from 36 g C m⁻² d⁻¹ to 953 g C m⁻² d⁻¹ with an average of 330 g C m⁻² d⁻¹.

Fluxes of CH₄ started to increase immediately after starting the composting process with a peak on October 20 (DOY 294) at 24.43 g C m⁻² d⁻¹. There was a CH₄ release immediately after turning the windrow on October 30 (DOY 304), peaking at 131.87 g C m⁻² d⁻¹ on November 1 (DOY 306). Then another peak of CH₄ occurred after the third turning of the windrow on November 6 (DOY 311) at 91.93 g C m⁻² d⁻¹; which declined rapidly and remained negligible after December 5 (DOY 340) for the remainder of the experiment.

Fluxes of N₂O fluxes began one day prior to the first turning of the windrow, and peaked the day of turning on October 22 (DOY 296) at 0.63 g N m⁻² d⁻¹. Prior to the second turning, N₂O fluxes began to rise again, but it was not until after turning the windrow there was a peak of N₂O flux that rapidly declined shortly thereafter. The highest peak occurred after the third windrow turning on November 8 (DOY 313) at 4.60 g N m⁻² d⁻¹. Fluxes of N₂O continued with variability

after the fourth windrow turning with a maximum of $2.71 \text{ g N m}^{-2} \text{ d}^{-1}$ on December 7 (DOY 342). Fluxes of N_2O gradually declined until they became negligible after December 30 (DOY 365). In the spring there was more N_2O fluxes that start on May 29 (DOY 515), reaching $1.42 \text{ g N m}^{-2} \text{ d}^{-1}$ on June 9 (DOY 526) and then became negligible after July 16 (DOY 563) to the end of the experiment.

Fluxes of NO occurred after the windrow was turned for the third time, and peaked on November 10 (DOY 315) at $0.32 \text{ g N m}^{-2} \text{ d}^{-1}$. Fluxes of NO were negligible thereafter for the remainder of the composting process. There were no observable trends in NH_3 or NO_2 measured, fluxes were negligible or negative in value throughout the composting process.

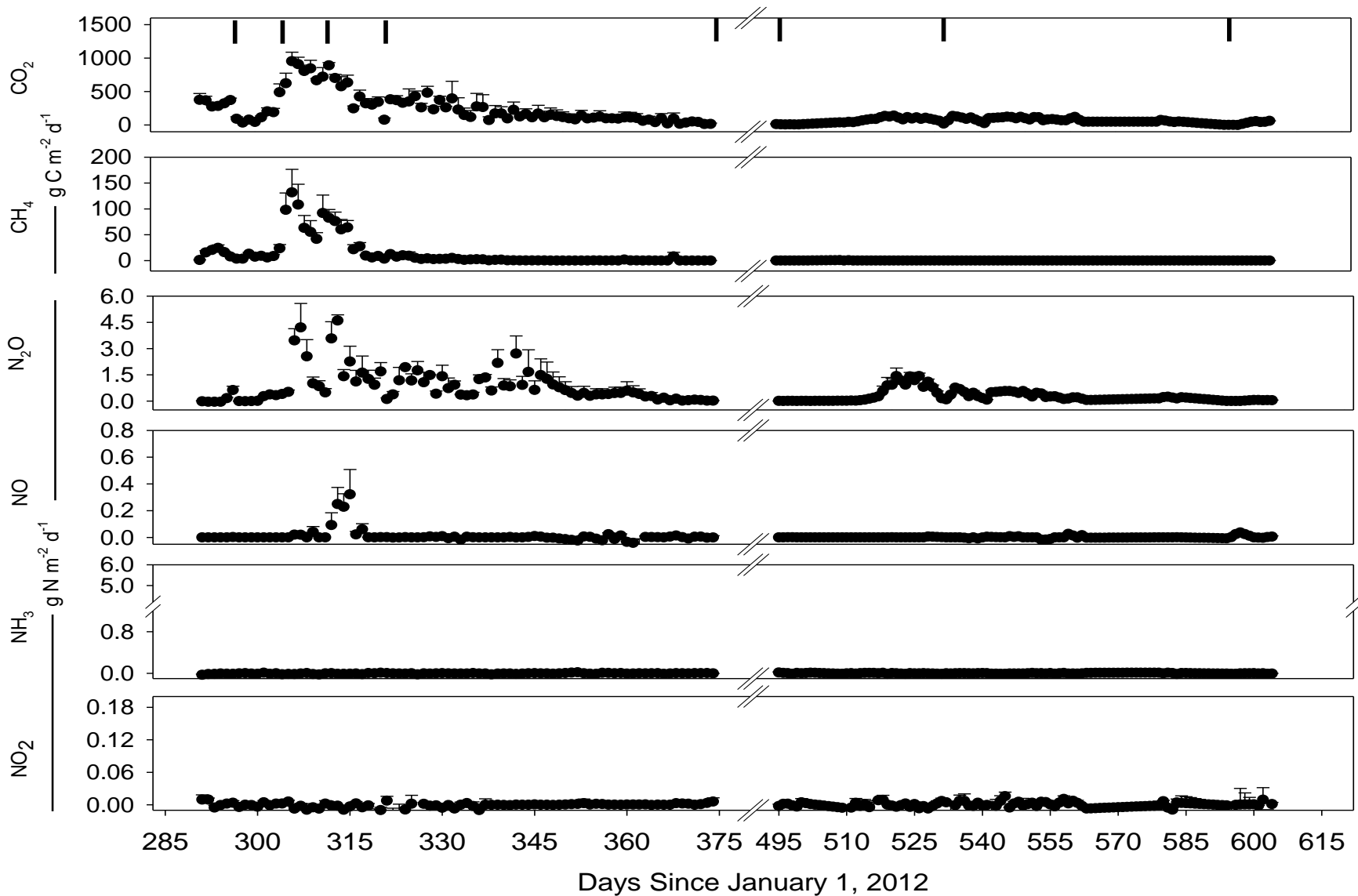


Figure 4.5 Daily mean gas fluxes and where shown ± 1 standard error for the windrow with woodshavings as a bulking material from October 17, 2012 (DOY 291) to September 6, 2013 (DOY 615). Bars indicate windrow turning.

4.4.3.3 Cumulative Gas Emissions

Gas fluxes were measured over 324 days for both the windrow with straw and woodshavings as bulking material. The windrow with straw as a bulking material had slightly higher but very similar cumulative CO₂ emissions on a per surface area basis compared to the windrow with woodshavings as a bulking material, 29,108 g C m⁻² and 28,567 g C m⁻² respectively (Table 4.3). However, since the windrow with straw as a bulking material had a greater exposed surface area, the total cumulative CO₂ emissions from the windrow with straw were double the amount compared to the windrow with woodshavings, 909,256 g C compared to 436,091 g C (Table 4.3). The total cumulative amount of CO₂ emitted per initial dry mass of the windrow was also almost double the amount for the windrow with straw, 632 g C Mg⁻¹, compared to 344 g C Mg⁻¹ for the windrow with woodshavings, because the weight of the starting material was similar (Table 4.4). Cumulative CH₄ emissions were almost double for the windrow with woodshavings, at 16.9 g C Mg⁻¹ compared to the windrow with straw, at 8.9 g C Mg⁻¹ (Table 4.4). Total mass losses for the windrow with straw and woodshavings, on a dry mass basis, were similar with 50% and 54% losses; similarly, total carbon losses were 64% and 66%, respectively (Table 4.4). However, measured emissions of CO₂ and CH₄ overestimated total C losses calculated by mass balance; the windrow with straw overestimated the C based emissions by 113% of the total C losses and the C based emissions for the windrow with woodshavings were overestimated by 14%.

For the windrow with straw as a bulking material, total gaseous N losses included cumulative emissions from N₂O, NO and NH₃, whereas the windrow with woodshavings only included N₂O and NO. Cumulative emissions of NH₃ was not included for the windrow with woodshavings as a bulking material because they were negative, similarly NO₂ emissions were also not included for either windrow. It is believed this is due to random variability during

calculation of the fluxes because fluxes were negligible and not that the windrow had not adsorbed NO_2 or NH_3 from the atmosphere; therefore, it was unnecessary to include them in cumulative emission calculations. Despite the lack of NH_3 emissions from the windrow with woodshavings as a bulking material, total cumulative gaseous N emissions measured were higher compared to the windrow with straw as a bulking material 1,524 g N and 1,057 g N, respectively (Table 4.3). Expressed on initial dry mass basis cumulative N emissions were almost double from the windrow with woodshavings, 1.2 kg N Mg^{-1} , versus the windrow with straw, 0.7 kg N Mg^{-1} (Table 4.4). This was due to the considerable amount of N_2O emissions measured from the windrow with woodshavings, 1.19 kg N Mg^{-1} of initial mass on a dry basis, accounting for 99% of the total N emissions. Whereas in the windrow with straw, N_2O emissions were 0.47 kg N Mg^{-1} and accounted for 64 %, in addition to NH_3 emissions of 0.19 kg N Mg^{-1} , accounting for 26% and NO emissions of 0.07 kg N Mg^{-1} , accounting for 10 % of the total gaseous N emissions. The windrow with woodshavings as a bulking material had less cumulative NO emissions at 0.01 kg N Mg^{-1} , accounting for 1% of the total N emissions measured. Total N losses calculated by mass balance were 0.7 g Mg^{-1} and 1.7 g Mg^{-1} for the windrow with straw and woodshavings, respectively, which resulted in losing only 4.7% and 12.9% of initial total N (Table 4.4). This indicated gaseous emissions overestimated total N losses calculated by mass balance by 6% for the windrow with straw as a bulking material, but only accounted for 69% of the total N losses in the windrow with woodshavings.

To compare direct greenhouse gas emissions, only cumulative emissions of CH_4 and N_2O are considered, using global warming potential (GWP) factors of 28 for CH_4 and 265 for N_2O , cumulative emissions expressed as CO_2 equivalents were 1,126 kg Mg^{-1} for the windrow with woodshavings as a bulking material compared to 526 kg Mg^{-1} for the windrow with straw as a

bulking material (IPCC 2013: GWP100 without inclusion of climate-carbon feedbacks) (Myhre et al., 2013). For the windrow with woodshavings as a bulking material, 56 % of the GWP was from CH₄ emissions and 44 % was from N₂O emissions; similarly, CH₄ accounted for a higher proportion of the GWP in the windrow with straw with 63 % from CH₄ emissions and 37 % from N₂O emissions.

Table 4.3 Cumulative gas emissions measured based on surface area and total amount measured as emissions from windrows with straw and woodshavings as a bulking material.

Gas	Straw	Woodshaving	Straw	Woodshaving
	g C or N m ⁻²		g C or N	
CO ₂	29,108	28,567	909,256	436,091
CH ₄	443	1,208	12,791	21,436
N ₂ O	26.59	101.16	679	1,505
NO	3.86	1.03	107	19
NH ₃	5.87	-0.34	271	-6
NO ₂	0.05	-0.17	0	-2

Cumulative emissions were calculated based on a span of 324 days. Total emissions measured (g C or N) were calculated based on estimated surface area at the time of emission.

Table 4.4 Initial, final and total losses of mass, total carbon (TC) and total nitrogen (TN) calculated by mass balance expressed as initial mass of the windrows as well as cumulative gas emissions measured expressed as initial mass of windrow. Losses of mass, TC, and TN are expressed as a percentage of initial mass, initial TC, or initial TN respectively as well as cumulative gas emissions expressed as percentage of the initial TC or initial TN, as a percentage of the TC or TN losses, and as a percentage of total C or N gaseous emissions measured for windrows with straw and woodshavings as a bulking material.

	Mass	TC	TN	CO ₂	CH ₄	N ₂ O	NO	NH ₃
	kg Mg ⁻¹			kg C or N Mg ⁻¹				
Straw								
Initial	1,000	470	14.6					
Final	499	170	13.9					
Total Loss or Emissions	501	300	0.7	632	8.89	0.47	0.07	0.19
				Percent (%)				
Of Initial Mass, TC, or TN	50.1	64.0	4.7	134.5	1.9	3.2	0.5	1.3
Of Total C or N Losses				210.4	3.0	67.6	10.8	27.4
Of Total C or N Gaseous Emissions				98.6	1.4	63.9	10.2	25.9
Woodshavings								
Initial	1,000	480	13.5					
Final	465	163	11.8					
Total Loss or Emissions	535	317	1.7	344	16.92	1.19	0.01	N/A
				Percent (%)				
Of Initial Mass, TC, or TN	53.5	66.1	12.9	71.7	3.5	8.8	0.1	N/A
Of Total C or N Losses				108.4	5.3	68.3	0.9	N/A
Of Captured Gaseous Losses				95.3	4.7	98.8	1.3	N/A

Mass, total carbon (TC), and total nitrogen (TN) are on dry mass basis. Values expressed as Mg⁻¹ are based on initial dry mass of the windrow. Did not include NO₂ in calculations because values were negative due to random variability, also did not include NH₃ in the windrow with woodshavings as a bulking material.

4.4.4 Final Compost Composition

The final compost produced from straw and woodshavings as bulking materials had CO₂ respiration values, a maturity indicator of the CCME Guidelines for Compost Quality, of <0.01 and 1.41 mg CO₂-C g OM⁻¹ day⁻¹, respectively (Table 4.5). Both types of compost produced had Solvita test values of 8 (Table 4.5). The C:N for the compost produced from woodshavings as a bulking material was statistically higher, at 18, compared to 12 for the compost produced from straw (Table 4.7). There were still a few small straw pieces visible in the final compost product from the windrow with straw as a bulking material, whereas the woodshavings were indistinguishable in the final compost product (Figure 4.6). The compost produced from woodshavings also had statistically higher NH₄⁺-N, whereas straw had statistically higher NO₃⁻/NO₂⁻-N values (Table 4.7). The ratio of NH₄⁺-N:NO₃⁻/NO₂⁻-N was 0.01 and 0.03 for compost produced from straw and woodshavings as a bulking material, respectively.

The compost produced by straw or woodshavings as a bulking material contained no foreign materials (Table 4.5). The composts produced had trace element concentrations below the maximum allowable concentrations; therefore, met criteria for CCME Guidelines for Compost Quality Category A compost that has no restrictions for application when used as per directions (Table 4.6). The composts produced had pathogen levels below what is allowable for CCME Guidelines for Compost Quality criteria (Table 4.5).

There were statistical differences in agronomic relevant parameters including nutrient values of Total N and Total K. The compost produced from woodshavings as a bulking material had lower total N, 21.6 g kg⁻¹, compared to the compost produced with straw, 29.3 g kg⁻¹ (Table 4.7). The compost produced from woodshavings also contained a lower concentration Total K, 4.6 g kg⁻¹ compared to straw with 14.6 g kg⁻¹ (Table 4.7) There was no statistical difference in total P

between the two types of compost produced (Table 4.7). There was not a statistical difference between pH; both composts produced had a pH value of 6.1 (Table 4.7). Final EC measurements were statistically different with values of 3.2 mS cm⁻¹ and 1.8 mS cm⁻¹ for the compost produced with straw and woodshavings as bulking materials, respectively (Table 4.7). However, total Na was not statistically different between the types of compost produced at 3.7 g kg⁻¹ and 3.8 g kg⁻¹ (Table 4.7), straw and woodshavings as bulking materials, respectively, but % base saturation of Na was higher in the compost produced from woodshavings at 4.32% compared to 3.41% for compost produced with straw (Table 4.8).

Organic matter for these two types of compost was not statistically different, with values ranging from 607 g kg⁻¹ for the compost produced from straw to 628 g kg⁻¹ for the compost produced from woodshavings. The BD of the compost produced was not statistically different at 399 kg m⁻³ for the compost produced from straw and 429 kg m⁻³ for the compost produced from woodshavings (Table 4.7). The particle size analysis indicated the compost produced from woodshavings is a finer material, with 84 % passing through the ¼ inch screen compared to only 46 % of the compost produced from straw passed through (Table 4.5). The MC is statistically different; the compost produced from straw had a higher MC at 705 g kg⁻¹ compared to the compost produced from woodshavings at 687 g kg⁻¹ (Table 4.7). The recommendation from A&L laboratories for the use of the compost produced from straw was to apply it for general landscaping, soil mix blending, and mulching and the compost should be blended with 2-3 parts soil, whereas the compost produced from woodshavings was recommended for use in general landscaping, soil mix blending, and light top dressing, based on the smaller particle size, and the compost should be blended with 3-4 parts soil.

Table 4.5 Pathogen, particle size, and maturity parameters that were analyzed for compost produced from straw and woodshavings as bulking material sampled on September 6, 2013 (DOY 615).

Treatment	Pathogens			Particle size				Maturity	
	Escherichia. Coli —— MPN	Fecal Coliform g ⁻¹ ——	Salmonella. P-A 25g(ml) ⁻¹	1 inch	½ inch	⅜ inch	¼ inch	CO ₂ Respiration CO ₂ -C gOM ⁻¹ day ⁻¹ mg CO ₂	Solvita Index
Straw	< 3	< 3	Negative	100	78.4	65	45.6	< 0.01	8
Woodshavings	<3	88	Negative	100	97.4	92.3	83.8	1.40	8
CCME (2005)		< 1000	< 3 MPN 4g ⁻¹					< 4	

Values indicated for CCME (2005) is maximum allowable to meet criteria to be considered Category A compost with no restrictions for use. P-A is a Presence-Absence Test.

Table 4.6 Trace element content of compost produced from straw and woodshavings as bulking material sampled on September 6, 2013 (DOY 615).

Treatment	Arsenic	Cadmium	Cobalt	Chromium	Copper	Mercury	Molybdenum	Nickel	Lead	Selenium	Zinc
	ug g ⁻¹ (dry basis)										
Straw	< 1	< 1	1.51	12.38	294.95	< 0.1	3.38	6.72	5.68	1.10	436.25
Woodshavings	< 1	< 1	1.25	10.53	363.30	< 0.1	3.35	5.47	6.49	1.30	519.50
CCME (2005)	13	3	34	210	400	0.8	5	62	150	2	700

Values indicated for CCME (2005) is maximum allowable concentration within product for Category A compost with no restrictions for use.

Table 4.7 Final characteristics of compost produced from straw and woodshavings as bulking material.

Treatment	BD kg m ⁻³	C:N	pH*	EC* mS cm ⁻¹	MC*	Total N	Total P	Total K	Total Na	Total OM	NH ₄ ⁺ -N*	NO ₃ ⁻ /NO ₂ ⁻ -N*
Straw	399(6) ^a	12(0) ^b	6.1(0) ^a	3.2(0) ^a	705(3) ^a	29.3(0.1) ^a	15.9(0.2) ^a	14.6(0.2) ^a	1.5(0) ^a	647(3) ^a	47(2) ^b	3,623(52) ^a
Woodshavings	429(11) ^a	18(1) ^a	6.1(0) ^a	1.8(0) ^b	687(2) ^b	21.6(1.0) ^b	16.2(0.8) ^a	4.6(0.2) ^b	1.6(0.1) ^a	680(12) ^a	56(1) ^a	1,840(15) ^b

Parameters marked with an asterisk (*) were sampled on September 6, 2013 (DOY 615), parameters without an asterisk were sampled on August 16, 2013 (DOY 594). Values with parenthesis are the mean of three samples \pm 1 standard error. Values for bulk density (BD) and moisture content (MC) are reported on an as-is basis and those for Total nitrogen (N), phosphorus (P), potassium (K), sodium (Na), organic matter (OM), ammonium (NH₄⁺-N), and nitrate/nitrite (NO₃⁻/NO₂⁻-N) on a dry weight basis. Means within the same column followed by the same lowercase letter (a, b) are not significantly different at P<0.05 ANOVA Tukey Test.

Table 4.8 Final cation exchange capacity and base saturation of cations of windrows with straw and woodshavings as bulking material sampled on September 6, 2013 (DOY 615).

Treatment	CEC		Base Saturation		
	meq 100g ⁻¹	% K	% Mg	% Ca	% Na
Straw	47.1	19.65	48.76	23.08	3.41
Woodshavings	38.2	7.06	57.49	27.99	4.32



Figure 4.6 Samples taken from windrows with woodshavings as bulking material (top left) and straw as bulking material (top right) at the start of the composting process October 15, 2011 (DOY 289) as well as samples taken from windrows with woodshavings as bulking material (bottom left) and straw as bulking material (bottom right) of the finished product on September 6, 2012 (DOY 615).

4.5 Discussion

4.5.1 Effect of bulking material on the process of composting slurry solids during winter conditions in Manitoba

It has been observed that weather conditions do affect the composting process when composting in outdoor windrows. For example, in Chapter 3, hot, dry, and windy conditions evaporated water from the windrows, slowing microbial activity at the beginning of the composting process and requiring water to be added which leached some of the slurry solids from the windrow. In this experiment, the weather was cool with high precipitation during the thermophilic stage of composting. Additionally, the area was sheltered from the wind and may have reduced evaporation of water from the windrows. The above average precipitation during the active stage of the composting process made it unnecessary to add water, therefore did not observe the slurry solids “wash off” the straw bulking material as in the previous experiment described in Chapter 3. Pre-soaking the bulking materials with water prior to adding the slurry solids was effective to maintain good contact between the slurry solids and bulking material and avoided leaching of slurry solids.

Starting the composting process in mid-October allowed enough time to turn the windrows once per week during the thermophilic stage to meet CCME Compost Quality Guidelines PFRP requirements (turning once feedstock materials were initially mixed plus an additional 4 turnings) before air temperature became too cold. The lower average temperature during the thermophilic stage of composting for the windrow with woodshavings as a bulking material compared to the windrow with straw as a bulking material was likely due to the anaerobic conditions. Less heat is produced by anaerobic microorganism compared to aerobic organisms, since the core temperature of the windrow with woodshavings was lower than 20 cm further from the core it suggests that severe anaerobic conditions were occurring at the core (Environment Canada, 2013).

In this experiment, after turning the windrows on January 8 (DOY 374), when air temperatures were -10 °C and the windrow temperatures were mesothermic, the windrows were heaped in a pile reducing exposed surface area and covered with a fabric geotextile cover. Instead of an abrupt decline in compost temperature it continued to gradually decline in temperature over a number of days, but still froze entirely. The fabric cover may have helped block the wind and added insulation to keep the cold air temperatures from freezing the windrow immediately. The cover also shed water and prevented snow from melting into the windrow to avoid further increasing the moisture content. This is an improvement compared to the previous experiment, described in Chapter 3, when the windrows were turned during the winter and there were only mesothermic temperatures occurring within the windrow, all activity stopped and the piles froze immediately which initiated a premature curing stage. However, prior to the windrows freezing entirely, there was no improvement in insulating against the winter temperatures by using woodshavings as a bulking material instead of straw, the decline in compost temperature started to occur around the same time. The bulk density of the windrow with straw as a bulking material was similar to that of the windrow with woodshavings as a bulking material by the time freezing temperatures occurred, providing similar porosity and insulation. The location of the windrows for this experiment was also sheltered from the wind better than in the previous experiment. The windrows in both Chapters 3 and 4 were relatively small to what might be expected in practice due to the lack of available slurry solids when they were needed to start the experiments. Piling more volume or combining windrows after the active stage of composting might improve the ability to keep temperatures above freezing throughout the winter.

After turning in the spring and throughout the summer the core temperature of the windrow with woodshavings as a bulking material generated mesothermic temperatures, which increased

the overall temperature average of the windrow and reduced surface temperature fluctuation from the air temperature. This also indicates the windrow with woodshavings took longer to mature compared to the windrow with straw.

4.5.2 Effect of Bulking Material on Relationships Between Composting Properties and Greenhouse Gas and Nitrogen Gaseous Losses

Fluxes of CO₂ that were emitted throughout the composting process aligned with the windrow temperature for both windrows. The active stage for the windrow with woodshavings as a bulking material was shorter in duration and less intense with a lower daily CO₂ flux average compared to the windrow with straw as a bulking material. During the active stage, CO₂ fluxes increased immediately after the windrow with straw was turned. Turning replenished oxygen and mixed nutrients. In the windrow with woodshavings as a bulking material CO₂ fluxes only increased after the second and third turning during the active stage. Since moisture content was not a limiting factor on CO₂ fluxes for either windrow, reduced CO₂ fluxes from the windrow with woodshavings could have been due to reduced availability of carbon and anaerobic conditions (Environment Canada, 2013). Hao et al (2004) measured CO₂ emissions from straw and woodchip bedded manure and found reduced CO₂ emissions from woodchip bedded manure due to slower decomposition of the woodchips. In the following summer of this experiment, the windrow with woodshavings as a bulking material released CO₂ fluxes, mesothermic microorganisms could have been dominant and were able to access more difficult carbon materials such as lignin and cellulose at lower temperatures (Paul and Geesing, 2009). The cumulative emission of CO₂ measured in both windrows was higher in this study compared to those seen in Chapter 3, which used straw as a bulking material in all three Trials, this was expected because this study was longer and the area was sheltered from the wind to reduce underestimation of CO₂ fluxes. Total mass and Total C

losses calculated by mass balance were also less than those seen in Trial 1 and 2 in Chapter 3; this was also expected as the type of turner and change in sampling techniques limited losses due to removal of material. However, in this study the gaseous C losses measured accounted for more than the Total C losses calculated based on mass balance. Here were still potential inaccuracies for doing mass balance calculations since the final mass of the windrow was estimated through volume and BD instead of being weighed and it was also assumed that the results of MC, BD, and Total N for the samples that were taken were representative of the entire windrow.

Since there were low oxygen concentrations measured within both windrows it was expected to measure CH₄ fluxes. Fluxes of CH₄ in the windrow with straw as a bulking material were closely related to oxygen content and only occurred midway through the active stage of the composting process. There was a build-up of CH₄ within the windrow when there were also high CO₂ fluxes, indicating high activity of microorganisms depleting oxygen that was slowly released. For the windrow with woodshavings as a bulking material CH₄ fluxes occurred at the start of the composting process, oxygen concentration was low due to the lack of pore space, high moisture content and demand for oxygen from the microbial activity. Build-up and slow release CH₄ fluxes occurred as well as during turning events. After the demand for oxygen from microorganisms declined CH₄ fluxes stopped even though the moisture content remained high and the pore space was low. The windrow with woodshavings as a bulking material did have higher cumulative CH₄ emissions compared to the windrow with straw as a bulking material. The magnitude of daily CH₄ fluxes emitted in this study were greater than in Chapter 3, and closer to what was measured by Hao et al. (2004). The cumulative amount of CH₄ measured from the windrow with straw, 8.9 kg C Mg⁻¹ was the same as the amount measured by Hao et al. (2004) for both straw bedded and woodchip bedded manure composting; whereas this study the windrow with woodshavings had

almost double cumulative CH₄ emissions at 16.9 kg Mg⁻¹. The smaller particle size of the woodshavings did not allow oxygen to flow through the windrow, creating anaerobic zones and greater methane production.

Much like the Trials in Chapter 3, the windrow with woodshavings as a bulking material could have produced N₂O fluxes as a result of both denitrification and incomplete nitrification. Fluxes of N₂O started early in the active stage of composting and were correlated with windrow turning. This could indicate complete nitrification and then denitrification (Fukumoto, 2009; Fukumoto et al. 2003) occurred. As discussed in Chapter 3, nitrification can occur on the surface of the windrow where temperatures were cooler and oxygen was available, building up NO₃⁻-N prior to assimilating to organic N. When the windrow is turned the build-up of NO₃⁻-N is turned into the center of the windrow where oxygen was limited and N₂O can be produced as a by-product of denitrification. Previous studies that have recorded fluxes of N₂O later in the composting process attributed this to be the result of incomplete nitrification, this process occurs when temperatures are reduced and nitrification begins; however instead of assimilating to organic N there is a build up of NO₂⁻-N due to lagging NO₂⁻-N oxidizing bacteria populations, therefore NO₂⁻-N converts to N₂O when oxygen is not limited (Fukumoto and Inubushi, 2009; Fukumoto et al., 2003). Incomplete nitrification that produced N₂O could have occurred in the windrow with woodshavings and the windrow with straw late in the active stage of the composting process as well as in the summer mesothermic stage. During this time, the compost temperature had decreased and NO₃⁻/NO₂⁻-N values had started to increase, there was also no association with turning. In both windrows cumulative N₂O emissions contributed the greatest amount to N gases measured. Overall cumulative N₂O emissions were more than double in the windrow with woodshavings as a bulking material compared to the windrow with straw, due to greater magnitude and duration of N₂O fluxes

as well as additional N₂O fluxes in the summer. Cumulative emissions of N₂O for both windrows in this study were also greater than cumulative emissions of N₂O measured in Chapter 3 or Hao et al. (2004). In this study the cumulative gaseous emission of N₂O from the windrow with straw may have been overestimated since gaseous N emissions accounted for more than the Total N losses calculated by mass balance. Again, potential inaccuracies of mass balance calculation still existed and relied on the assumptions that the samples were representative of the entire windrow. The Total N loss calculated by mass balance in this study may have been underestimated as losses of only 5 % were reported, reduced N losses calculated through mass balance were expected through reduced leaching and reduced removal of material from the windrows compared to Chapter 3, but this was also on the low end of total N losses of 8 to 42 % reported by Larney et al. (2006). Therefore, another possibility for increased measurement of gaseous N₂O could be due to better capture of N₂O fluxes in a sheltered area compared to Chapter 3 and more frequent data collection compared to Hao et al. (2004). In the windrow with woodshavings as a bulking material, the high cumulative N₂O gaseous emissions were expected since the magnitude of N₂O fluxes due to denitrification in anaerobic conditions was much more than observed in Chapter 3. However, the cumulative N₂O emission measured was not unreasonable at 8.8 % as a percentage of initial total N, which was still below 9.27 % of initial total N calculated in Fukumoto and Inubushi (2009). In the windrow with woodshavings gaseous N losses did not fully account for the Total N losses calculated by mass balance, the windrow with woodshavings could have also lost N through the emission of N₂, which was not measured, due to the severe anaerobic conditions (Hao et al., 2004).

In the windrow with straw as a bulking material NO fluxes occurred at the very beginning of the composting process along with NH₃ fluxes. Likely NH₄⁺ was undergoing rapid nitrification and NO was produced when oxygen was present (Hao et al., 2004). NO fluxes also occurred later

in the active composting stage associated with N_2O fluxes, which can indicate NO production through denitrification (Hao et al., 2004). In the windrow with woodshavings as a bulking material NO occurred fairly early in the active stage of composting but not with NH_3 emissions, the increase in NO corresponded with N_2O , likely indicating NO production through denitrification. The cumulative NO emission from the windrow with woodshavings as a bulking material was similar to that measured in Chapter 3, whereas cumulative emission of NO from the windrow with straw was higher due to greater magnitude and longer duration of NO fluxes. Due to the higher moisture content of the windrow with straw in this study compared to the Trials in Chapter 3 it was unexpected that higher cumulative NO emissions were observed, Fukumoto (2012) suggests NO fluxes to be higher in dryer conditions. However, since this study was conducted in a more sheltered area it may have allowed for NO to be captured and measured more accurately. In this study as well as Chapter 3, fluxes of NO_2 were not measured from the windrows, it is possible NO could have oxidized to NO_2 in the atmosphere but was not captured using the chambers.

In the windrow with straw as a bulking material, the occurrence of NH_3 fluxes were very similar to what was observed in Chapter 3, NH_3 fluxes occurred at the very start of the composting process when NH_4^+ was high as well as a few days after turning the compost during the middle of the active stage of composting and occurred with CH_4 and CO_2 . The complete inhibition of NH_3 fluxes, as was seen in the windrow with woodshavings, during composting of manures is unusual, but the factors such as low pH, low porosity and high moisture content certainly did exist to reduce NH_3 fluxes (Hansen et al., 2006, Fukumoto, 2009, Environment Canada, 2013, Paul and Geesing, 2009). Cumulative NH_3 emissions from the windrow with straw were similar to Trial 2 in Chapter 3, and accounted for 1.3 % of initial N, which was within range, but on the low end, of those

reported in other studies 0.3 % to 34 % of initial N (Fukumoto and Inubushi, 2009; Hansen et al., 2006; Jiang et al., 2011).

Overall the windrow with woodshavings experienced greater total gaseous N losses compared to the windrow with straw, 1.2 kg N Mg⁻¹ versus 0.7 kg N Mg⁻¹, respectively. The limited amount of pore space provided by using woodshavings and created conditions that reduced NH₃ fluxes; however, those conditions also encouraged anaerobic conditions which resulted in exceeding amounts of N₂O emitted after turning due to denitrification. The windrow with woodshavings as a bulking material also emitted more greenhouse gases compared to the windrow with straw as a bulking material, expressed as CO₂ equivalents, were 1,126 kg Mg⁻¹ and 526 kg Mg⁻¹, respectively. Again, the limited amount of pore space provided by using woodshavings made it difficult to maintain aerobic conditions, which created conditions for both CH₄ and N₂O gases to be produced and emitted, especially when the windrow was turned. From the perspective of reducing greenhouse gas and nitrogen gaseous losses, straw is the preferred bulking material over woodshavings.

4.5.3 Overall Compost Quality

The compost produced from either straw or woodshavings met the CCME Guidelines for Compost Quality and are within range of agronomic parameters to be considered quality compost. The composts produced met the CCME Guideline for Compost Quality CO₂ respiration maturity indicator as well as other maturity tests including Solvita, C:N, NH₄⁺-N: NO₃⁻/NO₂⁻-N, and temperature. According to these tests a finished compost must have a CO₂ respiration rate of less than 4 mg CO₂-C g OM⁻¹ day⁻¹, a Solvita rating of 7 or higher (ranked out of 8), a C:N ratio below 20, and NH₄⁺-N: NO₃⁻/NO₂⁻-N below 10 (Wood End Research Laboratory Inc., 2000; Larney et

al., 2006; Bernal et al., 1998; CCME, 2005). Overall, $\text{NH}_4^+\text{-N}$ values of the final product were less than values in all trials in Chapter 3, and $\text{NO}_3^-/\text{NO}_2^-\text{-N}$ values were higher, the longer duration of the study further progressed the trend of dropping $\text{NH}_4^+\text{-N}$ and increasing $\text{NO}_3^-/\text{NO}_2^-\text{-N}$, but did not reduce the total concentration of N. Both composts produced were considered mature after 324 days of composting; however, the compost produced from straw as a bulking material may have been considered mature in the early spring despite having straw pieces still visible. The increased length of time and turning did not reduce the appearance of visible pieces; the final compost product from the windrow with straw as a bulking material was similar to the appearance of Trial 1 in Chapter 3. The straw pieces likely would not interfere with crop production, however the compost product may not be aesthetically pleasing to markets outside of agriculture applications without screening. The increase in compost temperature of the windrow with woodshavings after turning in the spring indicated that this compost did require more time to mature because the woodshaving material was more difficult to degrade compared to straw. Additional time required to produce mature compost may be problematic if the producer wanted to apply or sell the product to free up storage space.

The composts produced were both well below the maximum allowable levels for CCME Guidelines for Compost Quality parameters for trace elements, presence of foreign materials, and pathogens. The agronomic parameters also indicate that a quality compost product was produced and could have value in multiple soil applications if used appropriately. For example, nutrient ranges of compost vary for Total N between 8-16 g kg^{-1} , Total P between 1.3-4.4 g kg^{-1} and Total K between 4.1-12.4 g kg^{-1} (Paul and Geesing, 2009). The compost produced from either straw or woodshavings as bulking material had above average Total N and Total P, and Total K concentrations were within the typical range. The compost produced from woodshavings as a

bulking material did contain less K, because less K was provided from woodshavings compared to straw. The pH of the compost produced was low, and could be an indication of the presence of organic acids that were formed under anaerobic conditions during the composting process; however, mixing the compost with an alkaline based soil or additive could increase the pH to a preferred range, usually between 6.5-8 (Paul and Geesing, 2009). The EC was a measure of all the soluble salts including salts that provide nutrients such as potassium, magnesium, calcium, nitrates and ammonia, as well as sodium and chloride salts that can be toxic to plant roots, Epstein (1997) suggests anything greater than 5 mS cm^{-1} may lead to phytotoxicity. The EC in this study should not be a concern however, it was important to look specifically at the %Na base saturation because greater than 1 % base saturation of the media can be toxic to plant root systems causing germination and emergence issues (A&L Laboratories Inc., 2004). As noted both composts produced have a %Na base saturation greater than 1 % thus must be blended or incorporated with soil in order to avoid this issue. It is interesting that the relative values of EC do not correlate to relative values of total Na or %base saturation of Na. It is also interesting that the final EC measured in Chapter 3, 1.6 mS cm^{-1} to 2.0 mS cm^{-1} , was lower compared to the EC measured for the compost produced with straw in this experiment, perhaps indicated that more salts were leached from the windrows in Chapter 3. Generally organic matter in compost ranges from $300\text{-}700 \text{ g kg}^{-1}$ (Paul and Geesing, 2009), the two composts produced during this experiment were at the high end of this range and were also higher than the OM values of 423 to 429 g kg^{-1} measured in Chapter 3. Less inorganic material from the soil making up the composting pad was incorporated into the windrows in this experiment compared to Chapter 3 due to the different compost turner used. The composts produced have high moisture content for transporting to end markets; final moisture content should be as close to 400 g kg^{-1} to maintain the microbial populations but minimize the

weight. Reducing the pile size and further turning can be done to evaporate the moisture and reduce the moisture content.

4.6 Conclusion

Overall either bulking material straw or woodshavings can be used to compost slurry solids; however, if other carbon sources are available woodshavings should not be used as the only source of carbon. Both windrows were able to provide conditions for thermophilic microbial activity and insulation to maintain thermophilic conditions in order to meet CCME Guidelines for Compost Quality PFRP prior to the windrow freezing in the winter. However, neither the straw nor woodshaving bulking materials were able to sustain mesothermic temperatures throughout the winter. In cool and wet weather conditions the potential for water evaporation from the windrows was minimal and thus difficult to determine if the bulking material had an effect on reducing water evaporation. There was also no need to add water to these windrows during this experiment because the moisture content stayed relatively high, thus unable to evaluate if there was good contact between slurry solids and the bulking material to avoid the slurry solids leaching or “washing off” when water was added. The compost produced from the windrow with straw in this experiment appeared the same, in terms of visible straw pieces, as the Trial 1 windrow in Chapter 3, this information as well as the windrow temperature not increasing above ambient air temperature indicates a longer duration was not required, application of the compost could be done in the spring. Whereas the compost produced from woodshavings does take longer to fully mature compared to the compost produced from straw as bulking material, despite the lack of woodshavings being visible. This may be a consideration if available space was limited for

composting and if the timing of when compost was ready for markets needed to be aligned with when the user can apply the compost.

During this experiment conditions existed for GHG and N gaseous losses in windrows with straw or woodshavings as a bulking material. There were greater total GHG and N gaseous losses from the windrow with woodshavings as a bulking material. The majority of the GHG and N losses were a result of anaerobic conditions. To avoid anaerobic conditions improved pore space must be maintained, further studies could look at the effect of using a mixture of woodshavings and straw together as a bulking material or use woodchips instead of woodshavings. The late emission of N₂O in the windrow with woodshavings was evidence that it was important to measure N₂O emissions while the compost is in the curing stage to fully account for N₂O.

Conducting the experiment in a sheltered area may have increased the capability of the automated chamber and FTIR system to capture C based gases. Measured emissions of CO₂ and CH₄ overestimated losses of Total C calculated by mass balance for both of the windrows. Measured emissions of N based gases were slightly overestimated from the windrow with straw but underestimated for the windrow with woodshavings compared to Total N losses calculated by mass balance. Reported losses of total mass, Total C, and Total N were lower and more reasonable compared to those calculated in Chapter 3. Removing less material during sampling and turning events improved determination of mass balance calculations. However, measured gas emissions were not expected to overestimate Total C or N losses calculated by mass balance, because there was still a relationship between increased wind speed and reduction in gaseous emissions (Chapter 5). In order to compare measured gaseous losses against mass balance losses requires better accounting of material removed during sampling, turning, or leaching and weighing the final material produced instead of estimating based on volume and bulk density.

The type of bulking material had little effect on the quality of compost produced. Both types of compost produced as safe to apply to the soil based on being below thresholds set in the CCME Compost Quality Guidelines for maturity, presence of foreign material, concentration of trace elements, and level of pathogens present. Both types of compost produced also have favourable agronomic parameters for marketing or using the compost in a variety of soil applications. The compost must be blended or incorporated in soil to prevent salt toxicity to plant roots and to buffer the pH, but will also add nutrients, including N, P, and K as well as OM to the soil.

5.0 OBSERVED EFFECTS OF WIND WHEN USING AN AUTOMATED CHAMBER AND FTIR SYSTEM FOR MEASURING GASEOUS LOSSES FROM WINDROW COMPOSTING

5.1 Abstract

When measuring gaseous emissions from a windrow using an automated chamber and Fourier Transform Infrared spectroscopy system, there is potential for under estimation of the gas fluxes due to wind. Increased wind speed, lower density of the feedstock material, and perpendicular wind direction, were identified as potential factors that increase the underestimation of gas fluxes. Further analysis of the data could be conducted to try to remove the error by utilizing a wind correction factor or choosing to not use data if the wind speed is above a certain threshold. Setting up the experiment in a sheltered area reduces the extent of underestimation of gas fluxes and should be considered for future studies.

5.2 Introduction

The automated chamber and Fourier Transform Infrared spectroscopy (FTIR) multi-gas analyzer system was proven to be capable of measuring gas fluxes from turned compost windrows (see Chapter 3 and 4). Chapter 2 identified how the system could be set up and potential errors in data analysis. However, another source of error was also observed shortly after starting the first field experiment (Chapter 3) that is due to the design of the automated chambers. Wind movement through a compost windrow may cause reduction in gas concentration resulting in underestimation

in gas flux determination. This observation has been researched in the past when using static vented chambers. Xu et al. (2006) observed reduced gas concentrations due to wind causing change in pressure differentials within the chamber while measuring gas fluxes from soil. The wind changes the pressure within the chamber and can interfere with the diffusion of gases and concentration of gases; an increased effect was correlated with decreasing density of the soil. Sommer et al. (2004) also observed that wind blowing horizontally through a windrow might reduce the amount of gas emissions being captured in the chamber on top of the windrow. In previous studies flux chambers are placed along the top of the windrow, as what was done in Chapter 3 and 4, to capture the majority of gases that get emitted from the top of the windrow because windrow composting is known to rely on the “chimney effect” to provide air to the inner core of the windrow, air is taken in through the sides of a windrow and is emitted out the top through convection. However, if the wind moved air horizontally through the windrow this could replace air in the windrow that has high gas concentrations with air that has background concentrations, and result in reduced diffusion of gases to the chamber headspace and underestimation of the gas flux. With the capability to monitor gas fluxes nearly continuously it is possible to compare gas flux intensity versus half hour wind speed measurements to look at this interaction and identify what factors influence the effect.

Therefore, the objectives of this study were to:

- a) Observe wind speed and gas fluxes measured by the automated chamber and FTIR system.
- b) Identify if there were factors that may increase or decrease the effect of wind reducing gas concentrations.

5.3 Materials and Methodology

To determine if wind had an effect on gas flux determinations wind speed measurements were compared against carbon dioxide (CO₂) gas flux determinations that were taken during the field experiments presented in Chapters 3 and 4. Carbon dioxide was used as the comparative gas because CO₂ gas fluxes can be measured throughout the composting process. However, it is known that the rate at which CO₂ is emitted can be associated with the activity of the microbial population and generally CO₂ fluxes decreased throughout the composting process. In Chapter 3, gas flux measurements were taken from two compost windrows, referred to as Trial 1 and Trial 2. Trial 1 windrow was started on September 14, 2011 (DOY 256) and Trial 2 started on October 15, 2011 (DOY 388), the active period for these windrows was from September 19 (DOY 261) to December 13 (DOY 347) and October 17 (DOY 290) to January 10, 2012 (DOY 375), respectively (Chapter 3). In Trial 1 it was observed that daily CO₂ emissions were unexpectedly low from September 19 (DOY 261) to October 26 (DOY 399) so they were identified separately to determine if this was caused by the wind or the composting conditions. There were no on-site wind measurements taken during the study presented in Chapter 3, hour wind speed measurements were from a wind monitor (05103-10, R.M. Young Company, Traverse City, MI, USA) 3 m above ground height and located at a weather station approximately 15 kilometers south-west of the composting site at the Trace Gas Manitoba Research Site, Glenlea MB.

In Chapter 4, gas fluxes were measured from two windrows; one used woodshavings as a bulking material, the other used wheat straw. Gas flux measurements were taken from October 18 (DOY 292) to January 8, 2012 (DOY 373) and again May 8 (DOY 432) to June 21 (537), 2013 (Chapter 4). To reflect the different stages in microbial activity the windrow with woodshavings was split into seven time periods and the windrow with straw was split into eight time periods

based on trends in daily CO₂ emissions. Half hour wind speed measurements were monitored using a wind monitor (05103-10, R.M. Young Company, Traverse City, MI, USA) from a height of 1.5 m from the ground located on-site within 1 m of the compost windrows.

5.4 Results and Discussion

During the study in Chapter 3, wind speeds ranged from 0.03 to 13.29 m s⁻¹. Fluxes of CO₂ from the windrow in Trial 1 ranged from -1,515 to 36,812 g-C m⁻² s⁻¹, and fluxes of CO₂ from the windrow in Trial 2 ranged from -1,435 to 48,571 g-C m⁻² s⁻¹ for the windrow started on October 15. The rate of CO₂ flux varied depending on the conditions of the windrow as well as the micro-conditions present at each chamber. There was reduced variability and reduced rate of CO₂ fluxes when wind speeds were over 9 m s⁻¹ for both Trial 1 and Trial 2 (Figure 5.1 and 5.2). In Trial 1, there was no indication that the wind speed was higher during that period of time, September 19 (DOY 261) to October 26 (DOY 399), when CO₂ fluxes were observed as relatively lower, ranged from only -402 to 8,396 g-C m⁻² s⁻¹. Therefore, low moisture content (see Chapter 3) slowing the activity of the microorganisms is likely the reason CO₂ fluxes were reduced rather than increased wind speed.

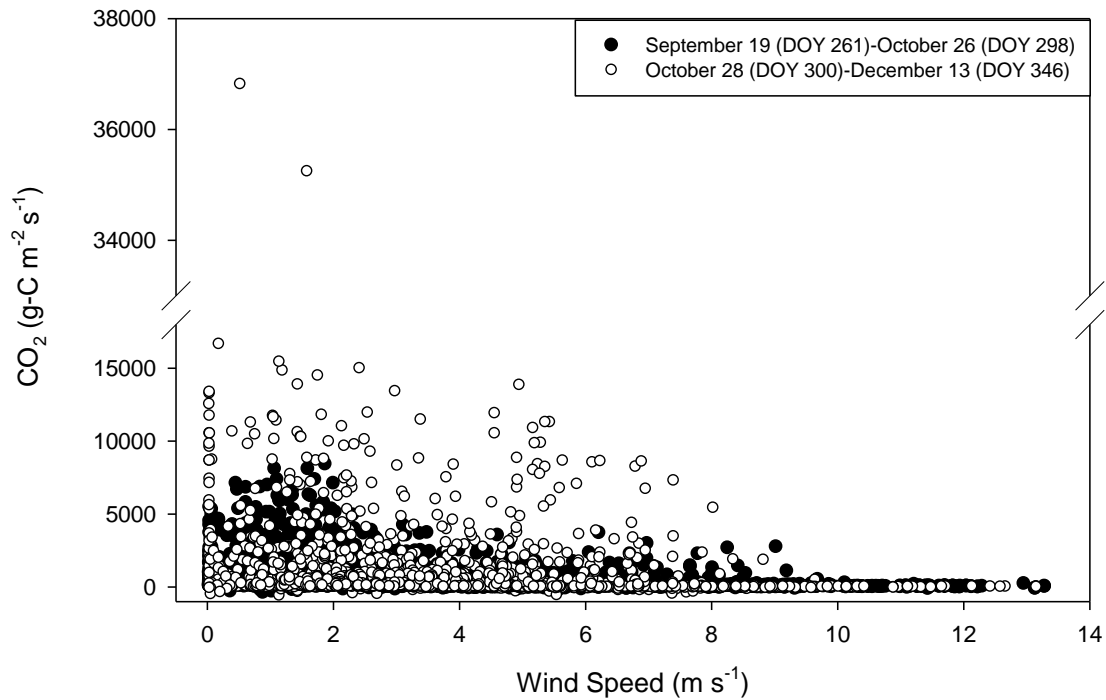


Figure 5.1 Carbon dioxide (CO₂) gas fluxes from chambers versus half hourly wind speed measurements during the active stage of the composting process for the Trial 1 windrow started on September 14, 2011 in Chapter 3.

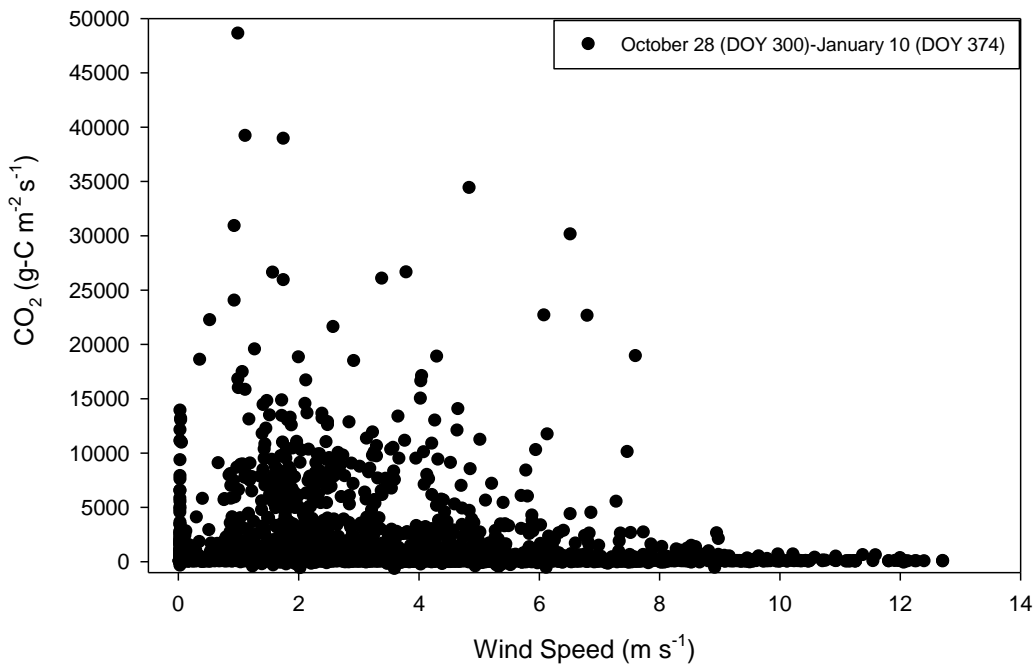


Figure 5.2 Carbon dioxide (CO₂) gas fluxes from chambers versus half hourly wind speed measurements during the active stage of the composting process for the Trial 2 windrow started on October 15, 2011 in Chapter 3.

During the study in Chapter 4, wind speeds ranged from 0 to 6.9 m s^{-1} , which was lower than what was measured in Chapter 3; possibly because the wind speed was measured next to the compost windrows in a sheltered area (Figure 5.3 and 5.4). It should be noted that the cumulative CO_2 emissions in Chapter 4 were actually overestimated, compared to what was estimated as carbon loss from mass balance, and the cumulative values were more than double compared to those in Chapter 3, that did not account for carbon loss from mass balance. Therefore, simply using a sheltered area to monitor gas fluxes reduced the wind speed and resulted in a reduced extent of underestimation of gas fluxes. Fluxes of CO_2 ranged from -683 to $25,816 \text{ g-C m}^{-2} \text{ s}^{-1}$ and -632 to $27,199 \text{ g-C m}^{-2} \text{ s}^{-1}$ for the windrows with woodshavings and straw, respectively. Carbon dioxide fluxes declined as the composting process went on because microbial activity decreased as the available carbon decreased and less CO_2 was generated. In the windrow with woodshavings as a bulking material the relationship between high wind speed and reduced gas fluxes is weak; the starting density of this windrow was higher than that of the windrow with straw as a bulking material. This could have prevented wind from flowing through the windrow. The windrow with straw as a bulking material shows a much stronger correlation between higher wind speed and reduced gas fluxes, when wind speed was above 2 m s^{-1} , variability and the rate of CO_2 fluxes were reduced. As the composting process progressed there was less of a correlation between wind and decreased CO_2 fluxes. Less CO_2 was being generated and the density of the windrow had increased.

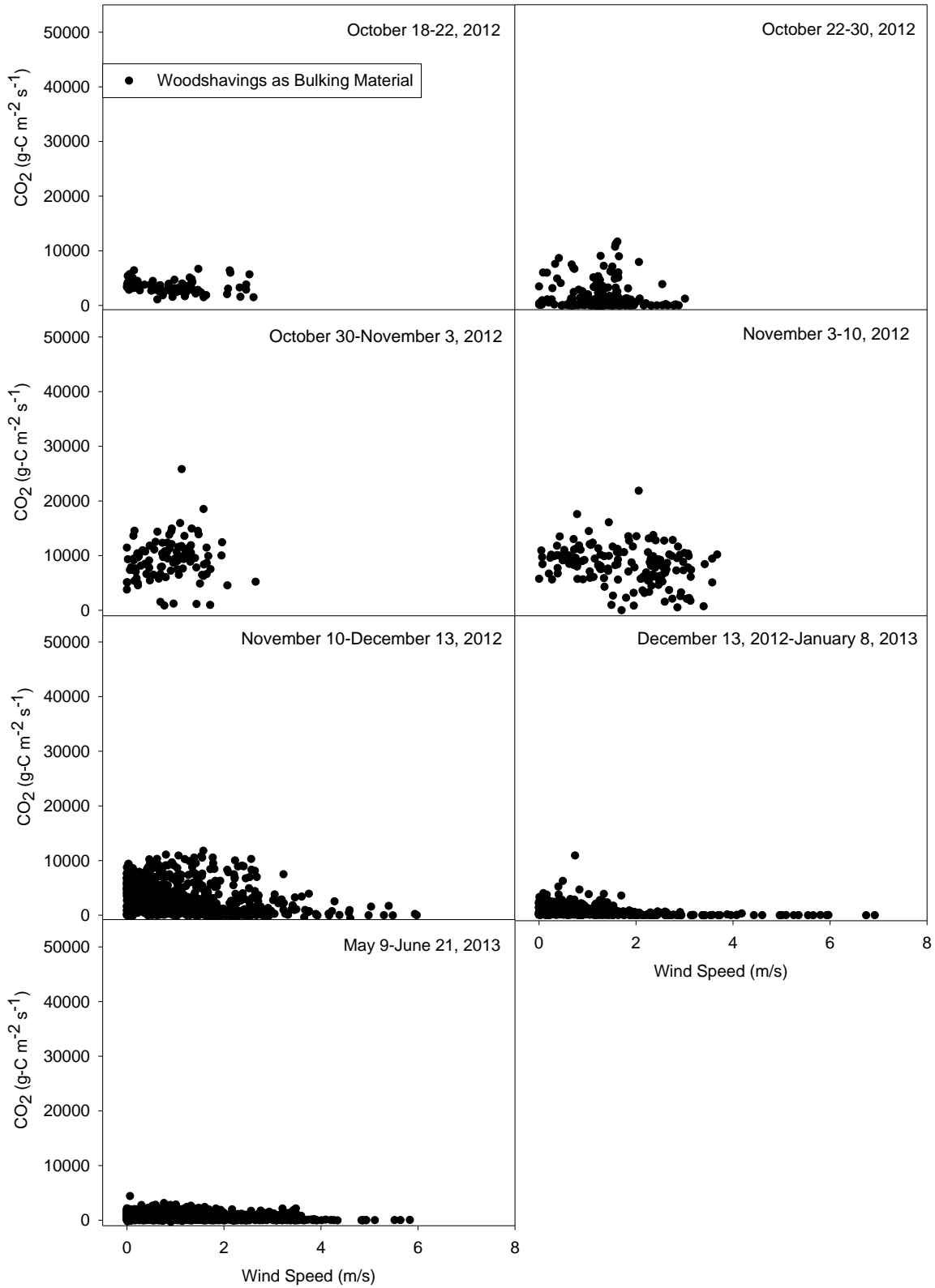


Figure 5.3 Carbon dioxide (CO₂) gas fluxes from chambers versus half hourly wind speed measurements throughout the composting process from the windrow with woodshavings as a bulking material in Chapter 4.

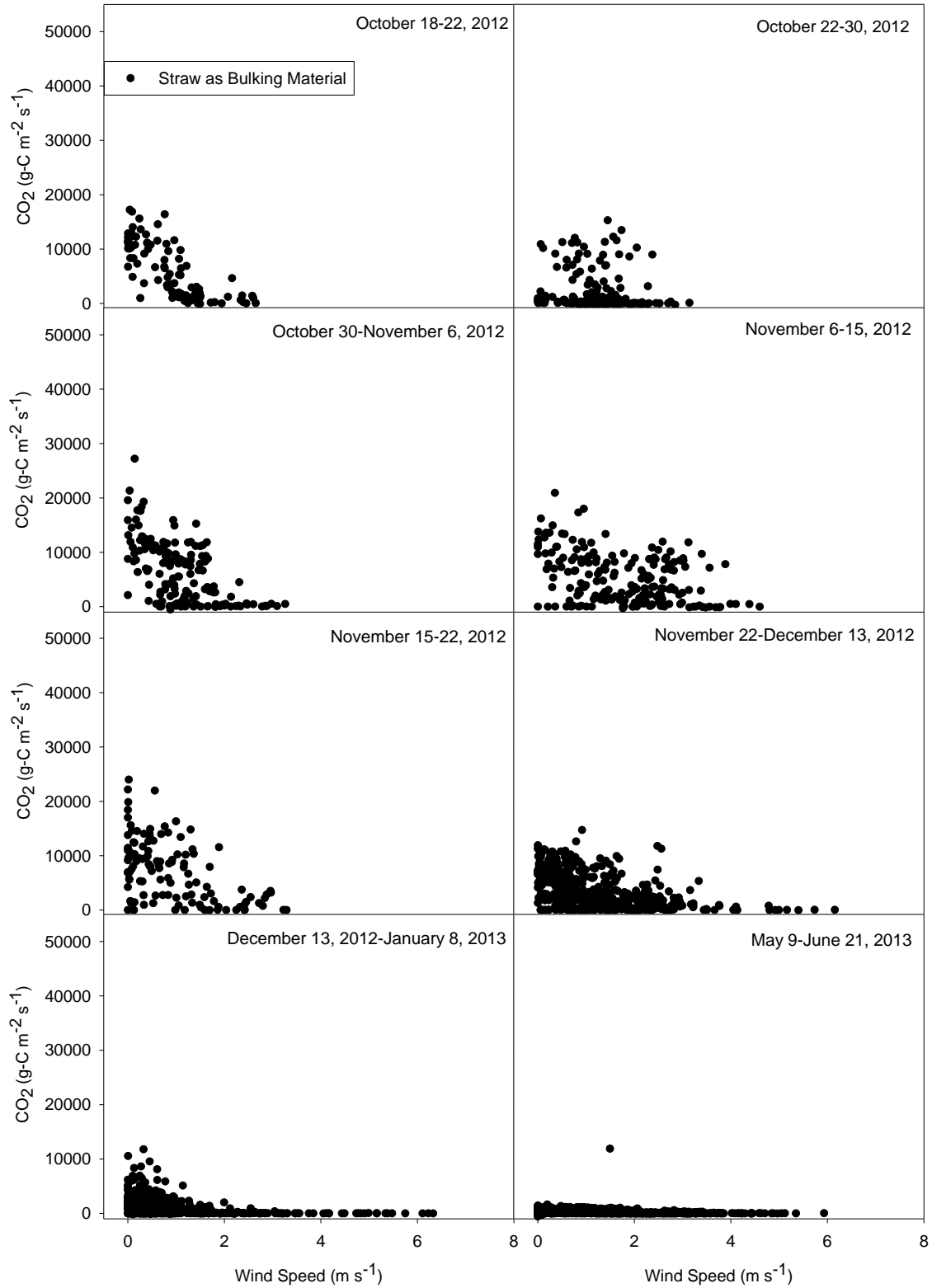


Figure 5.4 Carbon dioxide (CO₂) gas fluxes from chambers versus half hourly wind speed measurements throughout the composting process from the windrow with straw as a bulking material in Chapter 4.

The direction of wind may have also played a factor. Although it was not investigated, it is possible that the effect of wind was less if the direction of the wind is parallel to the windrow compared to if the direction of the wind is perpendicular to the windrow. The longer length of the windrow may slow down the wind resulting in less displacement of gas.

The amount of gas fluxes that may have been underestimated due to wind was not determined, but there appeared to be a relationship. Applying a wind correction factor after the data has been collected may be possible; factors that would have to be considered include wind speed, density of windrow, and wind direction. These factors change throughout the composting process and would have to be updated regularly. Another approach would be to set a wind speed threshold; gas fluxes that were measured at wind speeds above the threshold would be removed. The threshold would have to consider the density of the windrow, direction of the wind, the stage in composting process and any other micro-conditions present within chambers for example moisture content. Due to the intensive data analysis required, applying these approaches was beyond the scope of this project but could be performed in future studies. Fluxes measured by the automated chamber and FTIR system could also be compared against a different type of gas monitoring system to determine the extent gas fluxes are underestimated, similar to what was studied by Sommer et al. (2004) comparing an integrated horizontal flux and backwards Lagrangian stochastic dispersion micrometeorological techniques versus static vented chambers.

5.5 Conclusion

There was a relationship between increased wind speed and reduced CO₂ gas fluxes when using an automated chamber and FTIR flux system to monitor gases. Factors such as the speed and

direction of the wind as well as the density of the windrow have an influence on the extent of gas flux underestimation. However, the rate of gas production was also determined by conditions of the windrow, such as moisture content, and can vary by the stage of the composting process as well as micro-conditions of the chambers. A wind correction factor or threshold value could be applied, but would require extensive analysis of the data to account for the constantly changing factors. The automated chamber and FTIR system should be compared against another method for determining gas fluxes that considers wind as a factor such as dispersion micrometeorological techniques, in order to determine the extent of gas flux underestimation. Conducting the experiment in a sheltered area, such as in Chapter 4, may have reduced the extent of underestimation of gas fluxes and should be considered for future studies.

6.0 OVERALL SYNTHESIS AND RECOMMENDATIONS

6.1 Utilizing an Automated Chambers and the FTIR system for Assessing Gaseous Losses During the Composting Process

This study has been the first to demonstrate the ability of the automated chamber and Fourier Transform Infrared (FTIR) spectroscopy system for measuring gaseous fluxes from turned windrows during the composting process. There are many advantages to using this system compared to static vented chambers including frequent data collection, less possibility for human error, and the capability for multiple gas analysis including ammonia (NH_3), nitric oxide (NO), and nitrogen dioxide (NO_2) in addition to carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). The FTIR proved to be very capable of accurately measuring CO_2 , CH_4 , N_2O whereas measurement of NH_3 , NO, and NO_2 may have been less accurate because these gases are more reactive. Since NO_2 was not identified in either of the two studies, it may not be required to measure NO_2 during the composting process in the future. There was potential for interference between N_2O and CO_2 that resulted in an underestimation of N_2O . However, more investigation was required to determine if the extent of the error was relatively important given the high rate of N_2O fluxes from composting windrow compared to if the system were used in other applications such as measurement of N_2O from soil.

Sorption of NH_3 on the surfaces of the automated chamber and FTIR system and within water vapour was observed. To reduce condensation, it was recommended to keep the multiplexer in a heated and insulated shelter and change the Balston filters at least once per week when NH_3 was emitted. There was no difference in NH_3 sorption when using BevA or Teflon tubing to attach the automated chambers to the multiplexer, therefore, decided to keep the tubing as BevA to maintain flexibility.

Analyzing actual gas fluxes verified the use of an Excel Macro that was developed to select data to be used in a linear equation for calculating gas fluxes. The Excel Macro eased the use of data analysis especially for calculating CO₂, CH₄, and N₂O gas fluxes. However, a recommendation to further ease the use of the Excel Macro would be to use only one computer to run multiplexer and FTIR so that the time stamp on the chamber closing and the time on the FTIR analysis can be synchronized. When NO was emitted this required checking the data and sometimes manually changing the data points to have a linear fit. Potential errors in over or underestimating NH₃ fluxes due to sorption was observed; however, the majority of the NH₃ fluxes calculated did have a linear fit. To improve calculation of NH₃ gas fluxes it was recommended to extend the post-purge time to decrease the presence of residual NH₃ in the automated chamber and FTIR system. Using a different technique to measure gas fluxes, such as area under the curve, could also be considered for NH₃ in future studies.

The types and quantities of gases that were measured by the automated chamber and FTIR system could be associated and explained by the properties of the composting process and were relative to those measured in other literature. In the studies performed in Chapter 3 and 4 the gas fluxes both over and underestimated carbon and nitrogen losses as predicted by mass balance. Performing the study in an area sheltered from wind reduced the underestimation of gas fluxes. The quantification of mass loss and the mass balance calculations could be refined further than what was done in this study by conducting the experiment on an impermeable surface, weighing the final compost and recording the amount of material removed during sampling and turning. This may give a better comparison of gaseous emissions measured versus Total C and N losses calculated by mass balance. If these estimations were more comparable then there may be greater confidence in using the automated chamber and FTIR system. The automated chamber and FTIR

system should continue to be used as a research tool to compare management practices. The increased frequency of data collection over static vented chambers reduces the need for gap filling and can be necessary to capture the rapidly changing conditions and gases produced during the composting period. The ability to measure NH_3 and NO , in addition to N_2O , despite the challenges, gives an indication of the relative balance of N gaseous losses when comparing composting practices. Having the ability to measure multiple N gases could be useful for advancing the research in identifying the microbiology during the composting process, and could be used to manufacture high quality composts. It would be recommended to compare the performance of the automated chamber and FTIR system with a different gas monitoring system that factors in wind such as the integrated horizontal flux or the backwards Lagrangian stochastic dispersion micrometeorological technique to determine the extent of underestimation of gas fluxes due to wind.

6.2 Adding Value to Separated Pig Slurry Solids by Composting Throughout the Winter

Composting separated pig slurry solids allows for additional storage of the material throughout the winter and produces a marketable end product. Composting throughout the winter does present challenges due to the cold temperatures and addition of snow. Turned windrow composting can be a suitable technology if the turning was done frequently enough at the start of the composting process, while the windrow temperatures remain thermophilic, in order to meet the Canadian Council of Ministers of the Environment (CCME) Process to Further Reduce Pathogens (PFRP) criteria. If possible, after the active phase, the windrows should be combined for larger volume in winter to provide more insulation. Using woodshavings as a bulking material required less material but did take longer to fully mature compared to using straw, which may be a consideration for

selecting a bulking material if available space for composting is limited. Overall composting separated pig slurry solids produces a product that is mature and has consistent agronomic parameters that could be marketable. In order to sell the product to higher value markets, the compost would have to be screened to reduce particle size and remove any non-decomposed bulking material.

6.3 Best Management Practices for Reducing Greenhouse Gas and Nitrogen Gaseous Losses from Composting Separated Pig Slurry Solids

Composting separated pig slurry solids did result in greenhouse gas and nitrogen gaseous losses. In Chapter 3 and 4 all the windrows had a stage where high moisture content contributed to anaerobic conditions. As expected, anaerobic conditions increased the production of both greenhouse gases CH_4 and N_2O . It was important to have better control over moisture content, when possible the windrows should be sheltered from precipitation in a permanent structure or covered with breathable material that sheds water, such as the fabric geotextile cover that was used in Chapter 4. Generally, if composting throughout the winter there will be less evaporation occurring and precipitation added through snow, therefore the moisture content could be started at the low range of ideal moisture content. The woodshavings as a bulking material did not allow enough pore space for oxygen, which also contributed towards producing anaerobic conditions. For this reason, woodshavings should not be used as the only bulking material if other economic options are available. Future studies could monitor greenhouse gas and nitrogen gaseous losses from a combination of straw and woodshavings or woodchips as a bulking material. These options could be better at providing a heterogeneous mixture of particle sizes, availability of carbon, with the ability to adhere to separated pig slurry solids.

The concentration of total nitrogen in the final compost product stayed the same or increased from the initial product indicating nitrogen losses were not too excessive, but were actually consistent with mass loss. However, it would be ideal to observe an increase in nitrogen concentration. The majority of nitrogen gaseous losses were a result of N₂O emission. Nitrous oxide emissions could have occurred as a result of both denitrification under anaerobic conditions as well as incomplete nitrification. Therefore, it was important to continue measuring greenhouse gases through the curing stage to capture N₂O losses. NO contributed the smallest portion of nitrogen losses, which was expected since the moisture content of all windrows was high. Woodshavings were effective at reducing the emission of NH₃; therefore, woodshavings should continue to be considered as part of the recipe for composting separated pig slurry solids.

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APPENDIX A: EXCEL MACRO CODE FOR OBSERVATION FINDER

```
Sub obsfinder()
ConvertTime ("N2")
FindEndObs2 ("M2")
End Sub

Sub ConvertTime(StartCell As String) ' Start Cell is N2, string because its text
Dim Time As String '
Dim TimeRow As Integer
Dim Counter As Integer
Time = Format$(Time, "h:mm:ss") ' formats it to remove the DOY do we can perform calcs
TimeRow = Replace(StartCell, "N", "") 'Removes the N so we can work with integers
Counter = TimeRow + 1
While Range("N" & Counter - 1).Value <> "" 'Continue when the value of N does not equal 0
Range("M" & Counter - 1).Value = Range("N" & (Counter)).Value - Range(StartCell).Value
'Place the value of Ni-No in Mo
Counter = Counter + 1
Wend 'End while when StartCell is blank
End Sub

Sub FindEndObs2(StartCell As String)
Dim IVTime0 As Double
Dim TimePeriod As Double
Dim Counter As Integer
Dim NextMax As Double
Dim NextMaxAddress As Integer
Range("A2:B12,E2:K12").Copy
Cells(1, 16).Select
Selection.PasteSpecial
IVTime0 = Range("D12").Value
Counter = 2
While Range("M" & Counter).Value <> "" And Range("M" & Counter).Value > -1
TimePeriod = Range("M" & Counter)
NextMax = IVTime0 + TimePeriod
NextMaxAddress = 2
While Range("D" & NextMaxAddress + 1).Value <= NextMax
NextMaxAddress = NextMaxAddress + 1
Wend
Range("A" & NextMaxAddress - 10 & ":B" & NextMaxAddress & ",E" & NextMaxAddress -
10 & ":L" & NextMaxAddress).Copy
Cells((Counter - 1) * 12, 16).Select
Selection.PasteSpecial
Counter = Counter + 1
Wend
End Sub
```