Greenhouse gas emission during composting of different mixing combinations of natural resource by-products

By

Jayamini Rathnayake

A Thesis submitted to the School of Graduate Studies

in partial fulfillment of the requirement for the degree of

Master of Science

Boreal Ecosystems and Agricultural Sciences

Memorial University of Newfoundland

July 2023

Corner Brook, Newfoundland and Labrador, Canada

Abstract

Utilizing locally available natural resource by-products from forestry (wood shavings – WS, wood ash – WA, biochar – BC, and paper sludge – PS) and animal husbandry sectors (poultry manure – PM) to produce compost, may be an alternative for heavily using synthetic fertilizer and may help to improve soil physicochemical and biological characteristics. During composting, greenhouse gas (GHG) emissions are unavoidable due to microbial activities. The objectives of this research study were to: i) determine the best mixing ratio of PS with WS, and PM to reduce GHG (nitrous oxide – N_2O, carbon dioxide – CO_2, methane – CH_4) emissions during the composting process, ii) detect the impact of BC on GHG emission during composting of PS, WS, and PM, and iii) find out the effect of BC on quality of final compost product. There were four treatments with two mixing ratios; T1 – PS: WS: PM – 6:2:1 and T3 – PS: WS: PM – 1:2:1. In addition, T2 and T4 had 4% biochar added to T1 and T3, respectively. The high sludge mixing ratio showed higher GHG emissions than the low sludge mixing ratio for CO_2 and CH_4, but not for N_2O. The addition of 4% biochar did not show any significant difference among treatments for GHG emissions or the quality of the final compost although biochar showed some effects in reducing CH_4 and N_2O emissions. According to the final C/N, the compost did not reach its maturity within the 90 day period. However, all treatments reached a Germination Index of over 80%, indicating a reduction in phytotoxicity. Except for zinc and magnesium, all other micronutrients in all the treatments were within maximum acceptable limits for land application. Further studies are required to monitor GHG and NH_3 emissions in large-scale outdoor experiments using these treatments. Additionally, field trials are required to test the effect of compost on plant growth, which may help to further improve the final compost product.

General Summary

Agricultural expansion in Newfoundland and Labrador (NL) will increase the use of synthetic fertilizers, which may contribute significantly to greenhouse gas emissions. Locally, there are natural resource by-product waste streams that may be suitable as soil amendments, thus minimizing reliance on these synthetic agricultural inputs and reducing environmental risk. Paper sludge, wood ash, wood shavings, biochar, and poultry manure are renewable natural resource-based by-products derived from the province's forestry and animal husbandry sectors. Composting and using such by-products as agricultural soil amendments may result in enhanced soil quality and availability of nutrients. Composting is a natural biological process that can transform wet, smelly, bulky, and carbon-rich substrates into a nutrient-rich fertilizer. During the composting process, greenhouse gases, including nitrous oxide, methane, carbon dioxide, and ammonia are unavoidable. However, by mixing carbon-rich organic materials with other materials like biochar and nitrogen-rich waste by-products in correct combinations, it is possible to reduce greenhouse gas emissions during composting and increase the overall quality of the final product. The results of this work suggest that adjusting the components of the composting mixture have the potential to selectively decrease the emission of greenhouse gases while producing nutrient rich fertilizer.

Acknowledgments

I would like to express my sincere gratitude to my supervisor Dr. Mano Krishnapillai for all his support, motivation, and guidance to make this a successful project. His constant supervision and immediate feedback were invaluable during the research and writing of this thesis. Dr. Lakshman Galagedara, my co-supervisor, was encouraging me throughout this project and provided valuable comments, and suggestions. I would also like to thank my committee member Dr. Dmitry Sveshnikov for his guidance and support in finishing my research work. I would like to thank Dr. Tao Yuan, the lab coordinator in the BERI lab, for all his help with lab analysis. I want to extend my thanks to Dr. Yeukai Katanda, Ms. Sabrina Ellsworth (Director, Agriculture Production and Research Division, Agriculture and Land Branch, Government of Newfoundland and Labrador), Mr. Chris Pembroke (Manager, Environment and Technical Services, CBPPL) for all the support in supplying my experiment materials and help throughout this project.

I also acknowledge and appreciate the support provided by Dr. Mumtaz Cheema and Dr. Muhammad Nadeem during the research period. I would also like to thank my friends for their help and support during this degree, especially Sashini Pathirana and Kamsika Jeyarasa. Special thanks to the Office of Research and Graduate Studies of Grenfell Campus, Memorial University (MUN), and School of Graduate Studies MUN for providing financial support to complete my MSc. Degree. Mitacs Accelerate, Tourism, Culture, Industry and Innovation (TCII), Agricultural Clean Technology (ACT) program, and Corner Brook Pulp and Paper Ltd (CBPPL) are greatly acknowledged for their financial support in completing this research project. Finally, I would like to thank my family for their continuous support and encouragement.

Table of contents

List of Tables

List of Figures

List of Abbreviations

GHG = greenhouse gas

PS = paper sludge

WS = wood shaving

PM = poultry manure

BC = biochar

WA = wood ash

FW = fish waste

C/N = carbon to nitrogen ratio

OM = organic matter

GI = germination index

NL = Newfoundland and Labrador

CBPPL = Corner Brook Pulp and Paper Limited

EC = electrical conductivity

BERF = boreal ecosystem research facility

CHAPTER 1: Introduction

“The Way Forward” (Newfoundland and Labrador, 2016) was released in 2016 by the Government of Newfoundland and Labrador (NL) to provide planning for growth and sustainability in the agriculture of the province. Agricultural expansion in NL may lead to increased synthetic fertilizer usage and animal manure production, contributing significantly to greenhouse gas (GHG) emissions. Methane (CH_4) and nitrous oxide (N_2O) are considered significant GHGs contributing to the effect on global warming potential (Jiang et al., 2011). Carbon dioxide (CO_2) is also GHG, while ammonia (NH_3) leads to acid rain. The International Panel on Climate Change (IPCC) reports that the global warming potential of CH_4 and N_2O in the 100-year time frame is 25 and 298 times greater than that of CO_2, respectively (IPCC, 2007). This report also highlighted that about 13.5% of global anthropogenic GHG emission is from agricultural production and related industries (IPCC, 2007). High synthetic fertilizer usage, land-use practices, agrochemical usage, and manure and organic waste management practices in agricultural field are the sources of these emissions. Locally (NL), there are natural resource by-products and waste streams that may be utilized as soil amendments which may minimize the reliance on synthetic agriculture inputs. Paper mill sludge (PS), wood ash (WA), wood shavings (WS), and biochar (BC) are renewable natural resources based on by-products derived from the forestry sector. These products as well as animal industry-based by-products like poultry manure (PM), cow manure, and fish waste are readily available in NL.

The pulp and paper mill industry is one of the main contributors to the Canadian economy. During the production process, these facilities generate a large amount of organic waste, including paper mill sludge and biosolids. Canadian pulp and paper mills generate more than 1.5 million tons of paper mill sludge annually (Marche et al., 2003). PS generated from the pulp and paper industry may be used as soil amendments. Direct application has the potential to increase soil organic matter (OM) content, water holding capacity, soil pH, supply of soil nutrients, and decrease bulk density (Cabral et al., 1998; Manirakiza et al., 2021). However, direct sludge application to soil can have adverse effects, such as the decomposition of PS being very slow due to high cellulose and lignin content, it also can cause nitrogen (N) deficiency in the soil due to the priming effect (Turner et al., 2022). Also, nitrification during the decomposition process of PS can decrease the soil pH (Gagnon et al., 2012).

Corner Brook Pulp and Paper limited (CBPPL) produces about 10,000 kg of PS each day. The primary disposable method of this waste is incineration. As PS contains high water content (50−60%), burning it needs more bunker fuel or waste oil. Incineration of PS generates a high amount of carbon dioxide (CO_2), contributing to climate change. Since incineration is costly and produces toxic organic compounds harmful to the environment, CBPPL seeks different sustainable and environmentally friendly ways to recycle its waste (PS). Since CBPPL practices a mechanical separation process to extract pulp from wood, fewer contaminants (chemical and metal) remain in the waste produced during the process. Therefore, PS is suitable for uses like land application in agriculture and silviculture, composting, media amendments in agriculture, pyrolysis, and biogas production (Faubert et al., 2016).

Several studies have found that PS can be mixed with other organic materials and made into nutrient-rich compost (Hackett et al., 1999; Dinel et al., 2004; Hazarika and Khwairakpam, 2018). PS contains a high amount of lignocellulose; during the composting process, these materials are converted into more stable, nontoxic organic materials by microbial activities (Marche et al., 2003). As PS contains a high amount of carbon (C), incorporating nitrogen (N)-rich organic materials helps balance the C/N during the composting process, prevent soil N-immobilization, and increase readily available nutrient content (Faubert et al., 2016).

BC is produced by the pyrolysis of organic material and can be used as a soil amendment (Lehmann and Joseph, 2010). The application of BC improves soil pH and cation exchange capacity (CEC) and improves soil nutrient and metal retention (Yuan et al., 2011; Zhao et al., 2015; Fidel et al., 2017). Direct application of BC faces limitations due to its high cost and differences in properties (according to feed stock type and production conditions of BC, it can cause undesirable impact on soil microbial community, hydraulic characteristics, and nutrient retention). It is suggested that the co-application of BC with organic material can help to alleviate the restriction of using BC (El-Naggar et al., 2019). Also, several studies have shown that BC has an effect on mitigating GHG and NH_3 emissions during composting due to its porosity and adsorbent properties (Yin et al., 2021).

Different organic materials have different C/N (Marche et al., 2003). Therefore, the mixing ratio of the raw materials in a compost pile will determine the C/N of the pile which is a crucial factor to start and maintain the composting process (Pare et al., 1998). Imbalance between C and N in the compost pile will cause a loss in organic C and N in gaseous form due to microbial activities (Chowdhury et al., 2014). During composting, as a result of the nitrification and denitrification processes, about 0.02 to 9.9% of total N is lost as N_2O, and due to the lack of aeration (supply of oxygen, O_2), 2.5% total organic C is lost as CH_4 (Jiang et al., 2016). Maintenance of correct C/N and aeration during composting may reduce GHG emissions. Adding WS as a bulking agent and C source may help to improve the aeration of compost piles; this has the potential to reduce NH_3 and GHG emissions (Chowdhury et al., 2014).

The main goal of this study was to explore the potential of locally available natural resource by-products (PS, WS, PM) to produce nutrient-rich compost while simultaneously reducing GHG and NH_3 emissions. The study aimed to test two specific hypotheses: 1) The identification of appropriate mixing ratios of raw materials for composting, with PS as the primary carbon source, WS as a bulking agent, and PM as the nitrogen source, could lead to a reduction in GHG and NH3 emissions during the composting. 2) Addition of BC as a porous absorbent component would further contribute to the reduction of GHG and NH_3 emissions during composting. To test these hypotheses, the research involved two phases: a composting incubation study and a subsequent composting bin experiment, both conducted on a laboratory scale with the following specific objectives.

I. To determine the best mixing ratios of PS with WS, and PM to reduce GHG and NH_3 emissions during the composting process.
II. To detect the impact of BC on GHG emission during composting of PS, WS, and PM.
III. To find out the effect of BC on the quality of the final compost product.

CHAPTER 2: Literature review

2.1 Composting

Composting is a naturally occurring process of microbial digestion under aerobic conditions with adequate temperature and moisture. Composting is a hygienic transformation of organic waste into plant-usable nutrients source. During this process, microorganisms perform the complex metabolic process to increase their microbial biomass in the presence of oxygen (O_2), N, and C.

During this microbial digestion of organic matter, they produce heat, various gases, and stable solid organic material substrate, which has plant-available nutrients, known as compost (Meena et al., 2021). Composting is the exothermic oxidation of biowaste organic matter (OM), which is transformed by chemoheterotrophic microorganisms via two main phases (Bernal et al., 2017), bio-oxidative phase and maturation phase. The bio-oxidative phase further contains three different phases (Keener et al., 2000; Bernal et al., 2009) as shown below.

1. Mesophilic phase I
2. Thermophilic phase
3. Mesophilic phase II (cooling)
4. Maturation phase (curing)

Bio- oxidative phase

2.1.1 Mesophilic phase I

During this phase, the initial composting process starts at room temperature, and within a few hours or days, the temperature rises to 45°C. The utilization of N and C by microbes for their body assimilation results in an increase in temperature (Ayilara et al., 2020). This act is conducted by a heterogeneous group of microorganisms. Mesophilic bacteria, fungi, actinomycetes, and protozoa initiate the composting process. The decomposition of soluble compounds like sugar is high, which leads to producing more organic acids, and the pH of the compost piles can drop during this phase.

2.1.2 Thermophilic phase

When the compost pile temperature reaches higher than 45°C, mesophilic microorganisms are replaced by thermophilic microorganisms (organotrophic bacteria mainly). They have the capacity to grow at higher temperatures, like 65°C. These microorganisms facilitate the degradation of complex organic materials (cellulose and lignin). Thermophilic microorganisms degrade cellulose, hemicellulose, fat, and some lignin; during this phase, OM degradation is maximized (Bernal et al., 2017). During this phase, thermophilic microorganisms convert N into ammonia resulting in a rise in pH. An increase in pH causes the transformation of organic acids into CO_2 and CH_4 by microorganisms (Bernal et al., 2017). High temperatures during this phase help kill pathogenic microbes and weed seeds in the compost pile. Microorganisms oxidize OM to provide energy for their cell activities and growth. This oxidation of OM produces carbon dioxide (CO_2) and water vapor (H_2O) (Bernal et al., 2017). During this period, aerobic microbial digestion occurs for a longer period, resulting in the production of a large quantity of CO_2 and methane (CH_4) (Bernal et al., 2017). During this period, ammonia (NH_3) emission was also observed due to increasing temperature and pH, resulting in an increase in the ratio of NH_3 and NH_4^+(Szanto et al., 2007). Movement of warm air to the top of the compost pile helps to transport and release of NH_3 to the surface of the compost pile. During the thermophilic phase, due to high-temperature nitrification being insignificant, there is no or little nitrate (NO_3^-) production (Bernal, 2009; Himanen and Hänninen, 2011). Since NO_3^- is the substrate for producing nitrous oxide (N_2O), the latter is not produced in large quantities in this phase.

2.1.3 Mesophilic phase II

After the degradation of C and N sources, the temperature of compost piles reaches about 40–45°C. During this phase, polymer degradation continues, and fungal growth can be seen by the naked eye. When the temperature goes below 40°C, the pH of the pile decreases slightly as mesophilic organisms (actinobacteria and fungi) resume their activities. They degrade the lignocellulosic material, cellulose, and hemicellulose (Bernal et al., 2017). During the thermophilic phase, when temperatures are above 45°C, these mesophilic organisms survive as spores or dormant forms and start to activate after the temperature drops (Tuomela et al., 2000). Gradual decomposition of organic C results in a decrease in C/N and shows no N loss through gaseous emissions (Bernal et al., 2017). Also, during this phase, available NH_4^+ will start the nitrification process, and if denitrification also happens at the same time, a significant amount of N_2O emissions may occur (Brown et al., 2008). Also, if there is an O_2 deficiency in the compost pile, a large amount of CH_4 emissions can happen (Brown et al., 2008).

2.1.4 Maturation phase

The temperature of the compost pile drops down to ambient temperature. Reheating does not occur, and microbial stabilization and OM humification (formation of humus) take place (Bernal et al., 2009; Chen, 2003). Also, condensation of carbonaceous compounds and polymerization happens during this phase which helps in the formation of fulvic and humic acid. Stabilization of compost occurs; most of the mineralized N is nitrified to NO_3^-. The ideal moisture content during this phase is 45–50%, and the length of the phase is normally around four weeks (Bernal et al., 2017).

2.2 Compost maturity

Maturity is the degree of completeness of composting after curing. Compost physical characteristics like color and odor give a general idea of the decomposition stage reached, but it gives little information about the degree of maturation. A matured compost should not have a negative effect on seed germination or plant growth which implies the stable OM content and absence of phytotoxic compounds such as ammonia, ethylene oxide, organic acids, phenols, salts and heavy metals (Hue and Liu, 1995; Bernal, 2009; Barral and Paradelo, 2011). Compost maturity is connected with the support for plant growth or phytotoxicity (Bernal et al., 2017). It is related to compost microbial activity since phytotoxic substances are produced by microorganisms in unstable compost. Indicators that can be used to assess compost maturity are O_2 consumption and CO_2 production, enzyme activity, OM content, NO_3^- concentration, and outcomes of biological tests (plant growth tests, microbial biomass, and activity assessment) (Neher et al., 2022).

Composting is supposed to continue until the microbial transformation of OM is complete; the presence of a degradable compound indicates unfinished compost (Bernal et al., 2017). Water soluble C is the most degradable C fraction; therefore soluble C/N is used as one of the indicators to find compost maturity (Bernal, 2009). During the composting process, the C/N decreases as OM degrades, leading to an increased release of CO_2. Consequently, this process results in an elevated concentration of N.

Plant growth and phytotoxicity tests are biological methods for estimating the degree of maturity. Plant tests used for the quality standard can be divided into two categories: germination test and growth test to assess shoot growth and root biomass (Bernal et al., 2009). The germination test provides an instant estimate of phytotoxicity and how the growth of the seedling will be affected by the changes in the stability and maturity of the compost. Compost maturity tests based on phytotoxicity is a well-established popular method. The seed germination test (seed germination index – GI) was first suggested by Zucconi et al. (1981). GI is calculated by the radical length and germination percentage of the seeds in the sample compared to the control (Luo et al., 2018). The seed germination test consists of three main steps: prepare aqueous compost extract; incubate the seed with the extract; measure and calculate the indicator parameters such as germination percentage, seedling vigor index (SVI), relative seed germination (RSG) and relative radical growth (RRG) (Luo et al., 2018). Also, GI is correlated with some of the biological and chemical indices to evaluate compost quality. GI is positively correlated with humification while negatively correlated with the content of NH_4^+ (Tiquia et al., 1996; Guo et al., 2012). Therefore, seed GI is accepted as the most popular parameter to test compost quality.

2.3 Greenhouse gas and ammonia emission during composting

Rapid development and various human activities can cause the generation of a large volume of GHG emissions, which contributes to global warming and accelerates the global greenhouse effect. As a result, there is potential of the increasing occurrence of disease and pests, abnormal climate, frequent droughts, and sea level increase (Yin et al., 2021). Various studies have shown that global temperature will keep rising in the coming decades, which will affect the life of living beings on the planet (Hindman & Upadhyay, 2002). The common GHGs are CH_4, CO_2, N_2O, and NH_3. CO_2 is produced by the microbial decomposition of organic matter, and it plays a very crucial role in maintaining and balancing the heat of the earth; as a result, CO_2 affects climate change (Kweku et al., 2018). CH_4 is produced under anaerobic conditions by methanogens. Its greenhouse effect is 22 times higher than that of CO_2 (Yin et al., 2021). N_2O and NH_3 are produced in the N conversion process. N_2O plays a major role in depleting the ozone layer. Its effect is 296 times greater than the single molecule warming potential of CO_2 (Ermolaev et al., 2015). NH_3 does not have an effect on temperature increase, but it acts as the substrate for N_2O production, which makes it an indirect GHG (Yin et al., 2021). About 13.5% of global anthropogenic GHG emissions come from agricultural activities, constituting 60% of global N_2O, 50% of global CH_4, and less than 1% of global CO_2 emissions (IPCC, 2006). In 2016, net GHG emissions from Canadian agricultural activities was about 7% of Canada’s overall GHG emissions (Agricultural Greenhouse Gas Indicator - agriculture.canada.ca, 2021)

The utilization and application of organic material as amendments have become one of the main sources of GHG (Yin et al., 2021). Studies have found that the application of paper sludge (PS) in silviculture, in managed forest stands, significantly increased maximum soil CO_2 emissions by 90% and 126% compared to the mineral fertilizer treatment and control, respectively (Chuan-Kuan et al., 2005). CO_2 emission from PS decomposition is from biogenic sources, and it may not be included in GHG emission inventories. Literature on GHG emissions from land application of PS is barely sufficient to make decisions on the sustainability of the management practice. GHG emission assessments of PS composting using life cycle analysis are also rare (Faubert et al., 2016). Two studies done on the management of municipal waste and biosolids showed that composting can reduce or maintain similar GHG emissions compared with the landfilling of organic waste (Brown et al., 2008; Faubert et al., 2016).

Composting is a natural biochemical decomposition process. Its end product is valuable, but the process is associated with releasing gaseous emissions, including GHGs, into the atmosphere. These released GHGs are attributed to energy requirements for the operation of compost plants' and biochemical reactions within the organic waste (Sayara et al., 2020). Most of the organic C is converted to CO_2 (Hao et al., 2004). N_2O, CO_2, and CH_4 contribute to climate change, global warming, acidification, and eutrophication of ecosystems as a result of NH_3 deposition, which also contributes to the formation of particulate material in the air (Dhamodharan et al., 2019). GHG and NH_3 can be produced due to a lack of aerobic conditions, CH_4 is produced where anaerobic conditions happen, and N losses as N_2O and NH_3 are associated with the ammonification and denitrification processes during composting (Friedrich et al., 2011; Saera et al., 2013; Yang et al., 2013). But these GHG and NH_3 emissions are less than those generated from landfill and the waste-to-energy recovery process of OM (Sayara and Sanchez, 2021). During the composting process, the amount of emitted gases is highly influenced by the type of treated waste, compost technology, and operational conditions like aeration, which have a direct impact on reducing the rate of N_2O and CH_4 (Sayara et al., 2020).

There are two main ways of CO_2 emission during composting: biogenic and non-biogenic CO_2 emissions. Biogenic emission happens during biological degradation of OM at aerobic conditions and oxidation of CH_4 by aerobic methanotrophic bacteria during anaerobic digestion. This emission is the highest amount of gas generated during composting. Non-biogenic CO_2 emissions are from the energy and fuel consumption in the facility. The global warming potential of CH_4 is 25 times higher than that of CO_2 over the 100-year time horizon (IPCC, 2006); strict anaerobic conditions, neutral pH, and the presence of nutrient-rich organic matter substrates are ideal for the growing of methanogenic bacteria. During the early stage of composting, these conditions can be temporally found. According to the literature, most amounts of CH_4 emissions were recorded during the initial weeks of the process. At the beginning of the thermophilic phase, high temperature facilitates the reduction of O_2 solubility, which leads to the creation of anaerobic spots in a pile. N_2O and NH_3 are produced due to nitrification and denitrification (Nag et al., 2016). At the initial stages of the composting process, N is in the organic form. As the process continues, the mineralization of the organic N leads to forming of NH_3, which reacts with hydrogen ions and forms NH_4^+. NH_4^+ to NH_3 equilibrium is highly affected by the pH value and the temperature (Liang et al., 2006; Wang et al., 2013). NH_4^+ oxidizing bacteria and nitrite-oxidizing bacteria convert the part of the N to NO_3^- through the nitrification process; this nitrate under anaerobic conditions will further convert into N_2O (Sayara et al., 2020).

Compared to other organic waste management practices composting process produces less amount of GHGs and has less impact on global warming (Thyberg and Tonjes, 2017). Composting municipal solid waste produces about 41 kg CO_2 equivalent per tonne of waste, which is lower than the amount produced by landfilling (400 kg CO_2 equivalent per tonne) (Nordahl et al., 2020). However, when composting and vermicomposting were compared, it was found that vermicomposting process caused 78.1% lesser GHG emissions compared to composting process (Sayara et al., 2020). Composting method, average composting temperature, initial moisture content, initial total C and N content, pH, and aeration rate are the factors affecting GHG and NH_3 emission rates during composting (Sayara et al., 2020).

2.4 Paper sludge (PS) and complementary organic and mineral sources in composting

Paper sludge is generated from the pulp and paper mill industry as waste in their wastewater treatment plants. It is produced at two different stages in the treatment plant. In the first step, effluent passes through the primary clarifier, and primary sludge is recovered by the sedimentation process. After that top water in the primary clarifier is passed to the secondary clarifier, microorganisms are used to separate residues by digestion. Secondary sludge is thus also known as waste-activated sludge. After that, sediment from both primary and secondary clarifiers is sent for dewatering. At Corner Brook Pulp and Paper Limited (CBPPL), paper sludge (PS) is a mixture of both primary and secondary sludge. After dewatering, the moisture of the PS is reduced up to 60–70%. There are different pulp and paper manufacturing processes (mechanical, chemical, chemo-mechanical, and thermo-mechanical) (Faubert et al., 2016). According to the process, the quality and composition of the wastewater effluent properties change. CBPPL use thermo-mechanical separation for pulping, and their sludge lacks chemical contaminations. The final PS product is lignocellulosic material that is high in carbon (C) and poor in nitrogen (N). PS contains some heavy metals, which can be an environmental issue regarding their disposal (IPPC, 2001; Faubert et al., 2016). To apply PS as soil amendment, one must follow the regulatory standards to avoid soil and water contamination. In Canada, the heavy metal content of PS is usually less than that in the municipal biosolid and is within the regulatory limits of all the provinces (Faubert et al., 2016).

The majority of PS is landfilled or burnt for energy recovery (heat and electricity production) (Charest and Beauchamp, 2002). Application of PS for agricultural lands is also well-known to increase crop yield by improving soil fertility, nutrient content, pH, aggregates, water holding capacity, and cation exchange capacity (Faubert et al., 2016). Composting PS prior to land application increases its overall quality as an amendment (Joo et al., 2015; Faubert et al., 2016). Composting converts lignocellulose residuals into stabilized, nonhazardous material rich in humic-like substances devoid of pathogens through the biological digestion process (Hubbe et al., 2010). Composting PS with other N rich industrial waste by-products (municipal biosolids, animal manure, and chicken boiler floor litter) decreases C/N, prevents soil N immobilization, and makes PS amendment suitable for plant growth (Hubbe et al., 2010; Faubert et al., 2016). Beauchamo et al. (2002), composted de-inking (residual) PS with poultry manure (PM) and chicken broiler floor litter for 24 weeks with three initial N concentrations of 0.6%, 0.7%, and 0.9%. According to this experiment, the results showed that the composting mixtures with initial N content of 0.6% yield the lowest C/N after 24 weeks (Charest and Beauchamp, 2002). Initial pH and C/N were found to be the critical factors in the composting process; however, 24 weeks were not enough to produce fully matured compost. This shows the feasibility of composting PS by combining it with N sources. Composting of PS with animal waste and other N sources has produced a value-added soil amendment in other jurisdictions (Hazarika and Khwairakpam, 2018).

Animal manure is a valuable nutrient source that can be used to improve physical and chemical conditions of the soil (Munoz et al., 2004). PM contains a greater amount of N than other manures; 40–70% of that N is in the form of uric acid, which goes through microbial mineralization and makes ammonia (NH_3) (Nagumo et al., 2011). The nutritional composition of PM varies depending on the type of the bird, age of the bird, proportion of bedding to dropping, feed ratio, and manure handling system (Kulesza et al., 2022). The effectiveness of PM as a fertilizer has been doubted due to low C and the high potential of losing N and phosphorus (P) from leaching (Stiles, 2017). Composting PM prior to application potentially improves nutrient constituent ratios and produce more reliable N and P source that are readily available after composting (Stiles, 2017).

2.4.1 Fish pond waste as a N Source

Intensive aquaculture can also generate a significant amount of potentially useful waste, such as fishpond sediments. This fishpond waste (FW) can be utilized as a source of nutrients to apply as fertilizer. Composting can be applied as a method of managing FW on sites with available organic waste materials (Dróżdż et al., 2020). Studies have shown the feasibility of fishpond sediment as a valuable soil amendment that has high N content with C/N < 12 (Thunjai et al., 2004; Karak, 2013). The use of a similar pond waste from a “Tilapia” farm as a direct soil amendment has been shown to have a positive effect on N uptake and overall biomass production in lettuce (Radziemska et al., 2019). Due to lower C/N and relatively high-density fishponds, sediment can be used as a bulking agent to speed up the composting process with agricultural waste (Karak et al., 2013). Finally, fishpond sediments contain essential plant nutrients like N, P, and K (Karak et al., 2013).

2.4.2 Wood ash (WA) to amend the pH of compost

WA is produced as a by-product during pulp and paper mill operations. Fly ash and bottom ash together are considered WA which is generated at the bottom of the furnace during the high-temperature combustion of wood, bark, and other organic residues. WA is composed of various amounts of calcium, magnesium, sodium, silicon, manganese, and iron, with heavy metal contamination, and having a pH variation between 10 and 12 (Bougnom et al., 2009). Many studies have shown that WA can be used as a liming material to raise soil pH (Bougnom et al., 2009; Serafimova et al., 2011; Bang-Andreasen et al., 2017). Properties of the WA depend on plant species, plant parts used in the combustion process, the fuel source used in the process, and storage conditions of the ash (Someshwar, 1996). Most WA produced from industries is sent for landfilling (Amous, 1999). The effectiveness and product quality of compost are affected by the quality of the substrate (Ricardo et al., 2014). Acidity in substrate results from the early production of fatty acids (Sundberg et al., 2004). If the substrates used in the process are acidic, when the decomposition process starts, lactic-acid bacteria and the yeast increase, causing a pH drop during composting (Ricardo et al., 2014). This can result in a decrease in product quality and hygiene (Sunberg et al., 2004). Different authors suggest the addition of alkaline ashes as buffer material for pH reduction (Koivula et al., 2004; Kuba et al., 2008). According to the literature, the addition of WA to kitchen biowaste composting has enhanced the process by improving the mineralization rate, formation of humic acid, oxygen availability, and reduced and controlled malodor (Koivula et al., 2004). Newfoundland soils are acidic and thus require the addition of liming material to neutralize soil for agricultural use (Maheswaran et al., 2020). Adding compost made from natural resource by-products also helps to overcome this problem (Bougnom et al., 2009).

2.4.3 Wood shavings (WS) as a bulking agent

The composting process is more effective with the right C/N and moisture content (Chang and Chen, 2010). Bulking agents play an important role in maintaining moisture and C/N in compost piles, effective in controlling air supply, and other important parameters (Batham et al., 2013). There are different kinds of organic materials which are used commonly as bulking agents; wood shavings, sawdust, grass hay, wheat straw, corn stalks, grass clippings, leaves, and fruit and vegetable waste (Chang and Chen, 2010; Kulikowska et al., 2022). These bulking agents are used in the composting process according to the need of the compost mixture to control moisture, increase airflow and porosity, act as an energy source for microbes, and enhance the degradation of compost materials (Manish et al., 2013; Awasthi et al., 2015). One of the commonly used bulking agents is wood shavings which are a by-product of the forestry industry (Kulikowska et al., 2022). The shape and texture of WS provide plenty of space within the pile, increasing airflow and temperature production. When composting high moisture materials like PS and FW, it reduces the excess water by absorbing it, which helps to maintain proper moisture balance inside the compost pile. The main component of WS is C (60.8%), and the WS have a small amount of N (0.9%) (Aljameel et al., 2021). They have a high content of cellulose, lignin, and hemicellulose (Horisawa et al., 1999), which makes them take longer to breakdown. According to Fabrice et al. (2021) addition of WS helps to maintaining appropriate moisture, it reduces greenhouse gas (GHG) concentrations, improves the physicochemical parameters, arrests leachate, prevents groundwater pollution, and reduces air pollution in windrow composting of livestock manure.

2.5 Biochar (BC) and its effect on the mitigation of GHG emission

BC is a relatively stable carbonaceous material, that can be manufactured from different kinds of OM under high temperatures with moderate oxygen supply (Awasthi et al., 2020). BC has the ability to adsorb CO_2 and NH_3 and simultaneously increase porosity which helps to reduce anaerobic pockets inside compost piles, thereby reducing CH_4 and N_2O emissions; also, BC may have the ability to reduce the mobility of water-soluble organic compounds, decreasing their degradation and reduce C and N loss by GHG emissions (Zhang and Sun, 2014; Wang et al., 2017, Awasthi et al., 2020). BC properties depend on many factors, like pyrolysis conditions and substrate type (Awasthi et al., 2020). According to Rajkovich et al. (2012), plant-based BC has a larger surface area and can absorb a greater amount of NH_4^+, GHG, and NH_3 compared to BC made by manure pyrolysis.

Studies have shown that application of BC during composting can mitigate GHG emissions (Lin et al., 2022; Mao et al., 2019). BC has a more porous structure and large surface area, and abundant surface functional groups, which make it more effective in mitigating GHG emissions during composting process (Cha et al., 2016). BC is used as an additive to improve microbial activities in the humification process, and mitigate GHG emissions (Yin et al., 2021). These capabilities are mainly attributed to the unique physicochemical properties of BC.

Abundant pores in BC provide a high capacity for water holding and improve aerobic conditions inside piles by absorbing excess water (El-Naggar et al., 2019) and, as a result, improve the aerobic environment inside compost piles (Wu et al., 2017). A study done by He et al. (2018) found that granular BC improves porosity by 4.02% and decreases CH_4 emission by 22.15%. The large surface area of BC provides a suitable environment for microbial habitat and thus enhances microbial activity (Xiao et al., 2017). Also, BC increases the activities of cellulase and urease by 56% and 96%, respectively, by enhancing the overall activity of microorganisms (Yin et al., 2021).

Abundant functional groups like, e.g., carboxyl, hydroxyl, epoxy, carbonyl, acyl, ester, ether, amino, and acyl groups on the surface of BC act as adsorption sites to enhance the adsorption capacity of BC during composting (Xiao et al., 2018). In addition, these functional groups enhance the microbial degradation of organic compounds during composting (Yin et al., 2021). Also, it has been found that BC reduces N loss during the composting process by adsorbing NH_4^+ and promoting the nitrification of compost products. The alkaline pH of BC provides a suitable pH range inside the compost pile for microbial activities (Xiao et al., 2017) and inhibits the emission of CH_4 by affecting the pH of the compost pile (Mao et al., 2018).

2.5.1 CO_2 mitigation

Even though studies found that BC has an effect on CO_2 emission, those effects are inconsistent among different studies. One study found that the addition of BC during composting of pig manure reduced CO_2 emissions by 26.1% (Wang et al., 2018). But according to another study, the addition of BC increased CO_2 emission by 53.2% during pig manure composting (Mao et al., 2018). These variations may be explained by BC increasing the isolation of external organic matter when added to composting to reduce CO_2 emissions (Liu et al., 2017) or BC providing a better environment for microbial activities and accelerating degradation of organic matter to produce more CO_2 (Mao et al., 2018).

2.5.2 CH_4 mitigation

CH_4 production needs an anaerobic environment and low oxidation-reduction potential (Liu et al., 2017). BC improves porosity inside compost piles and reduces anaerobic pockets, and

BC also can change the oxidation-reduction potential by improving the permeability, which decreases the activity of methanogens and enhances the activity of methanotrophs (Yin et al., 2021). Also, BC absorbs NH_4^+, which reduces the N utilization by methanogens, thereby reducing CH_4 emissions (Yin et al., 2021). Also, the adsorption of NH_4^+ weakens the competitive inhibition of the methane monooxygenase activity and enhances the oxidation activity of methanotrophs, which further reduces CH_4 emission (Yin et al., 2021).

2.5.3 N_2O mitigation

BC reduces the amount of inorganic N that can be utilized by nitrifying bacteria and denitrification bacteria by adsorbing NH_4^+ and NO_3^-, thus making a reduction in N_2O emission (Xiao et al., 2017). BC can absorb N_2O and reduce it to N_2, and BC can also suppress the expression of the *nos*Z enzyme, which catalyzes N_2O production, thereby leading to reduced N_2O emission during composting (Xiao et al., 2017).

2.5.4 NH_3 mitigation

NH_3 and NH_4^+ are adsorbed by the acidic functional groups on the surface of BC (Yin et al., 2021). BC enhances the activity of nitrifying bacteria, which convert NH_3 to NO_3^- and retain N within the compost (Akdeniz, 2019). Also, BC increases cellulase activity and produces more dissolved organic C via the decomposition of cellulose, which enhances the microbial utilization of NH_4^+ and reduces NH_3 emissions (Yin et al., 2021).

2.6 Effect of BC on GHG emission under different conditions

Many studies have shown that adding BC can mitigate various GHGs during composting process (Chen et al., 2018; Awasthi et al., 2020; Yin et al., 2021). But the results showed the effect of BC varied greatly among studies due to different types of BC used during the experiment and the environmental conditions of application. The effect of BC on GHG emissions during composting has been studied under different conditions, such as BC type (Chen et al., 2017), various dosage (Awasthi et al., 2020), particle size (He et al., 2019), and pyrolysis temperature (Yuan et al., 2015; Yin et al., 2021). Also, some studies have shown that combining BC with additives such as zeolite and wood vinegar has more effect on mitigating GHG emissions (Wang et al., 2017; Wang et al., 2018).

2.6.1 Effect of BC type

Studies have shown that raw materials used to make BC have a significant effect on its quality for mitigating GHG emissions during composting (Chen et al., 2017; He et al., 2019). Abundant pores in BC structure improve the aeration of the compost pile, enhance microbial activities of aerobic bacteria, nitrifying bacteria and methanotrophs (but inhibit denitrifying bacteria), and the surface of BC adsorbs NH_4^+, NH_3, and N_2O (Lin et al., 2022). BC produced from crop residues and woody biomass is better at mitigating CH_4, NH_3, and N_2O emissions during composting (Lin et al., 2022).

A study done by Chen et al. (2017) using BC made from different plant material (bamboo, corn straw, coir, woody, and layer manure) on the CH_4 and NH_3 emissions support the above conclusion. According to Chen et al. (2017), BC made from corn straw showed the lowest CH_4 and NH_3 emissions since they had high surface to volume area. Also, a comparison between bamboo BC and rice straw BC on GHG emissions during composting showed that adding bamboo BC resulted in 19.79% and 42.01% lower CH_4 and N_2O emissions, respectively, as bamboo BC had larger pore volumes and surface area (He et al., 2019).

2.6.2 Effect of BC dosage

Many studies have shown that effect of BC on mitigating GHG emissions significantly differs with the dosage of BC added (Lin et al., 2022). In most of the studies, BC was added in the range between 0 and 20% (w/w), and the addition of about 10% was the most effective (Awasthi et al., 2017). According to a study done by Awasthi et al. (2017), the addition of 2% to 18% BC resulted in a significant difference in GHG mitigation and adding 12% BC had the greatest effect. The addition of 2–6% BC had little effect on the emission of GHG (Awasthi et al., 2017). Also, similarly, another study showed that the addition of an excessive amount of BC may have adverse effects on the composting process (Lin et al., 2022). A study done by Liu et al. (2017), adding 5, 10, and 20% BC in PM composting, found that the addition of 10% BC led to severe moisture losses and heat dissipation in the compost pile, which had a negative effect on the composting process. According to a study done by Wang et al. (2013), addition of 3% bamboo BC in manure composting reduced N_2O emissions. And composting of sewage sludge and wood chip with 4% of BC (w/w) showed an increase in CO_2 emission and a decrease in NH_3 emissions during the first week (Maliṅska et al., 2014).

2.6.3 Effect of BC particle size

Particle size significantly affects the pore characteristics and specific surface area of BC (Yin et al., 2021). A study done by He et al. (2018) showed that the addition of powdered BC increased CH_4 emission by 56.8%, while the addition of granular BC decreased CH_4 emission by 22.1%. Also, the addition of powdered BC was more effective in mitigating NH_3 emissions since more functional groups were exposed on the surface of the powdered BC, and the particle size of the BC had little effect on N_2O emissions (He et al., 2019).

2.6.4 Combining BC with additives

Combining BC with other additives can have a great effect on mitigating GHG emissions (Yin et al., 2021). A study done by Wang et al. (2017) showed that the addition of BC with zeolite during pig manure composting reduced the N_2O and NH_3 emissions by 78.13% and 63.4%, respectively, compared with only BC. Also, a similar study showed that mixing BC with bacterial powder during pig manure composting reduced CH_4, N_2O, and NH_3 emissions by 69, 45, and 26%, respectively, compared to only BC (Mao et al., 2019).

2.7 Effect of BC on the quality of compost

In addition, BC influences pile density and particle size, thereby facilitating oxygen (O_2) supply and avoiding the formation of anaerobic pockets inside the pile (Liu et al., 2017). Even though BC influences the initial pH of the compost pile, it does not significantly affect the final compost product pH (Maliṅska et al., 2014; Liu et al., 2017). Alkaline properties of BC and characteristics of compost substrate and compost process do affect the pH dynamic of BC amendment (Xiao et al., 2017). BC amendment affects the moisture content of compost mixture (Chowdhury et al., 2014) by increasing porosity, and BC can physically entrap water molecules in micropores and fissures (Xiao et al., 2017). This depends on the physical properties like surface area and surface tension of BC. The addition of oversized BC can lead to decrease moisture in compost piles due to the excessive pile porosity (Xiao et al., 2017). As a C-rich material additive, BC influences the C/N of the starting mixtures (Xiao et al., 2017). But BC takes decades to degrade, therefore when calculating the initial C/N of the mixtures, there is a need to consider the labile C rather than the overall C in substrates. Also, BC amendment favors N retention in composting, increases readily available P, and increases water-soluble K+, Na+, and Ca2+ content (Zhang et al., 2016; Chen et al., 2017; Awasthi et al., 2017).

The addition of BC influences microbial activities in the compost piles (Zhang et al., 2015; Sun et al., 2016; Xiao et al., 2017). By helping to develop porous structure with high surface area which supplies suitable habitat for microorganisms, and enhanced ventilation and moisture provides microorganisms with optimal living conditions inside the compost pile (Maliǹska et al., 2014; Zhang et al., 2014; Sun et al., 2016; Liu et al., 2017). Also, the addition of BC influences compost microbial community diversity (Awasthi et al., 2017; Chen et al., 2017).

CHAPTER 3: Methodology

3.1 Preliminary studies

Fresh fishpond waste (FW), collected directly from the salmon hatchery/nursery facilities in Stephenville, NL, was used as the nitrogen (N) source to balance the C/N with paper sludge (PS) and wood shaving (WS). Sediment and water samples collected from the bottom and top layers of the pond were homogenized to produce a uniform sample. Immediately after sampling, fish waste was capped in air-tight containers and stored at 4°C allowing the use of the same initial fish waste for all experiments. Consistency of moisture, pH, electrical conductivity (EC), total carbon (C), and total N was measured every time a new fish waste was used. Wood ash (WA) and PS samples were collected from Corner Brook Pulp and Paper Limited (CBPPL). WS was collected from Burton’s Cove Logging at Hampden, NL.

Figure 3.1 Raw materials used in the experiment: air dried poultry manure (PM) (A), paper sludge (PS) (B), biochar (BC) (C), fishpond waste (FW) (D), wood ash (WA) (E), and wood shavings (WS) (F).

Initial laboratory tests were carried out for all materials used for the experiment to measure moisture percentage, total C, total N, pH, EC, nutrient analysis, and C/N shown in Table 3.1. Samples were sent to the Soil, Plant, and Feed Laboratory of the Government of NL (NLSPF) in St. John's, NL, for analysis. Biochar (BC), produced by AirTerra Canada Company (Alberta), was used for the experiment. Poultry manure (PM) collected from a farm in Deer Lake, NL, was used as an alternative N source (Figure 3.1 indicates the raw materials used in the experiment).

Table 3.1 Initial analysis of raw materials before composting

Properties	PS	FW	PM	WS
Moisture (%)	65.5	92.5	67.8	8.7
pH	6.4	6.1	8.5	5.3
Total N (%)	1.36	4.91	5.46	0.038
Total C (%)	44.9	39.4	24.8	42.6
C/N	33:1	8:1	4.5:1	1121:1
Total phosphorus (P) (mg/L)	5400	53100	19000	< 50
Total potassium (K) (mg/L)	2200	1700	25100	660
Total calcium (Ca) (mg/L)	12200	103500	92300	760
Total magnesium (Mn) (mg/L)	1600	5600	4700	190
Total iron (Fe) (mg/L)	1350	2260	688	109
Total copper (Cu) (mg/L)	62	48	75	< 0.2
Total manganese (Mn) (mg/L)	2680	392	592	118
Total zinc (Zn) (mg/L)	374	1250	618	9
Total boron (B) (mg/L)	40	32	58	1.3
Soluble salts (dS/m)	0.33	18	-	2.3

A series of 24.6 L pail buckets with air-tight lids were used for the experiment. Buckets were specially designed and constructed to conduct forced aeration, monitor inside temperature, and extract gases collected inside. There were six treatments with three different mixing combinations of raw material with a target C/N of 26±2:1. BC was added to each combination to test the effect on GHG emissions. WA was added to test if it impacts the pH of composting mixtures, as fishpond waste has low pH.

The experiment includes three replicates of six treatments:

1. T1–FW+PS+WS+WA+BC → 40:1:7: 1:1
2. T2– FW+PS+WS+WA → 40:1:7:1
3. T3– FW+WS+WA+BC → 40:7:1:1
4. T4– FW+WS+WA → 40:7:1
5. T5– FW+WS+BC → 40:7:1
6. T6– FW+WS → 40:7

3.2 Method Development

The experiment was arranged in a completely randomized design (CRD) and carried out for 45 days under room temperature conditions at the Soils Lab, Grenfell Campus, Memorial University of Newfoundland (GC–MUN). During this period, the temperature was measured every day, and pH, EC, and GHG samples were collected every three days. The moisture content of the composting mixtures was measured once every two weeks.

The high moisture (92.5%) of FW made it challenging to handle and control the moisture of mixtures while balancing C/N. Continuous N loss due to volatilization during storage and even after mixing with other materials made it difficult to achieve targeted C/N. Because of that, in the second trial, an additional N source, *i.e.*, the PM, was added to the same mixing combinations of the first trial with targeted C/N of 40±2:1 and carried out an experimental trial to check microbial activity startup and temperature rise. Following this preliminary test, a few other tests were carried out with different mixing combinations to achieve different C/N (35±2:2, 45±2:1, 50±2:1, and 60±2:1) to find the best C/N for the temperature rise during the composting process. High C/N were targeted as available C for microbial use in PS, and WS is relatively low (Gagnon et al., 2003).

During the first and second trials, temperature rise, and notable microbial activities were not observed, which indicated that the composting process did not start. Since FW had some drawbacks like high moisture (more than 90%), N volatilization, and difficulty in handling, FW was removed from other trials done, and PM was used as the primary N source. Additionally, as an inoculating material, old compost was added on a weight basis ratio (20 g per 1 kg of mixture).

In a few trials with different C/N combinations of PM with varying rates of mixing of PS and WS at C/N of 38±2:1, temperature reached 50°C±5 in all treatments, which indicated microbial activities. Therefore, when planning final experiments, these findings were considered to decide on the treatments and their C/N.

3.3 Experimental setup

3.3.1 Incubation study

A small-scale incubation study was conducted for 90 days to observe microbial activities and measure GHG emissions during the process. One-L mason jars were used for the incubation study. As shown in Figure 3.2 and Figure 3.3, lids were pierced for aeration, and jars were covered with insulating materials to reduce temperature losses during the composting process.

Figure 3.2 Plan images showing 1 L mason jars filled with composting material with pierced lids.

In the jars, headspace was maintained at 300 mL. In addition, another set of lids was prepared with outlets to collect GHG and connect directly to the GHG-measuring instrument (Gasmet DX4015, Gasmet Technologies Oy., FI-01730 Vantaa, Finland). The PM was used as the primary N source with other C materials; PS, WS, WA, and BC. There were six different treatments. In all treatments, C/N was maintained at 40±2:1 (initial raw material analysis report for WS, PS, and WA was used for calculations). The PM was air-dried to bring the N to a constant level by reducing the volatilization of N and sent to the external laboratory to analyze the total C and N in the air-dried samples (these new values were used for calculation). BC was added at the rate of 4% by weight of the total compost mixture, and C content in BC (treatments 2, 4, and 6) was not considered when calculating total C/N, because C in BC is not readily available for microbial digestion. T2, T4, and T6 were the BC added treatments of T1, T3, and T5, respectively. The moisture level of mixtures was maintained at 60% (wet weight basis).

Experimental treatments carried out in three replicates were:

1. T1 – PS+WS+PM → 6:10:6

2. T2 – PS+WS+PM+BC → 6:10:6:0.5

3. T3 – PS+WS+PM → 11:6:4

4. T4 – PS+WS+PM+BC → 11:6:4:0.5

5. T5 – PS+WS+PM+WA → 6:6:3:1

6. T6 – PS+WS+PM+WA+BC → 6:6:3:1:0.5

Figure 3.3 Images showing 1 L mason jars filled with composting material with lids prepared for GHG collection (left) and jars covered with insulating materials (right).

3.3.2.1 Compost bin design

24.6 L pail buckets with airtight lids were used to design a special compost bin. It was designed in a way to collect GHG, monitor the temperature inside the bucket, supply forced aeration, and collect leachate from the bottom of the bucket (initially, as mentioned in the preliminary study description, FW was used as an N source since it has a high moisture level, it was expected to produce substantial amount of liquid leachate during the process). In this setup, bottom leachate collecting outlet was sealed (Figure 3.4) as FW was not used.

To monitor temperature variations inside the bin, a Digi-key TP 29-ND thermocouple probe was inserted in the middle of the bin. Forced aeration was supplied using Uniclife UL40 Aquarium Air Pump Dual Outlet connected to the diffuser, which was fixed to the mesh.

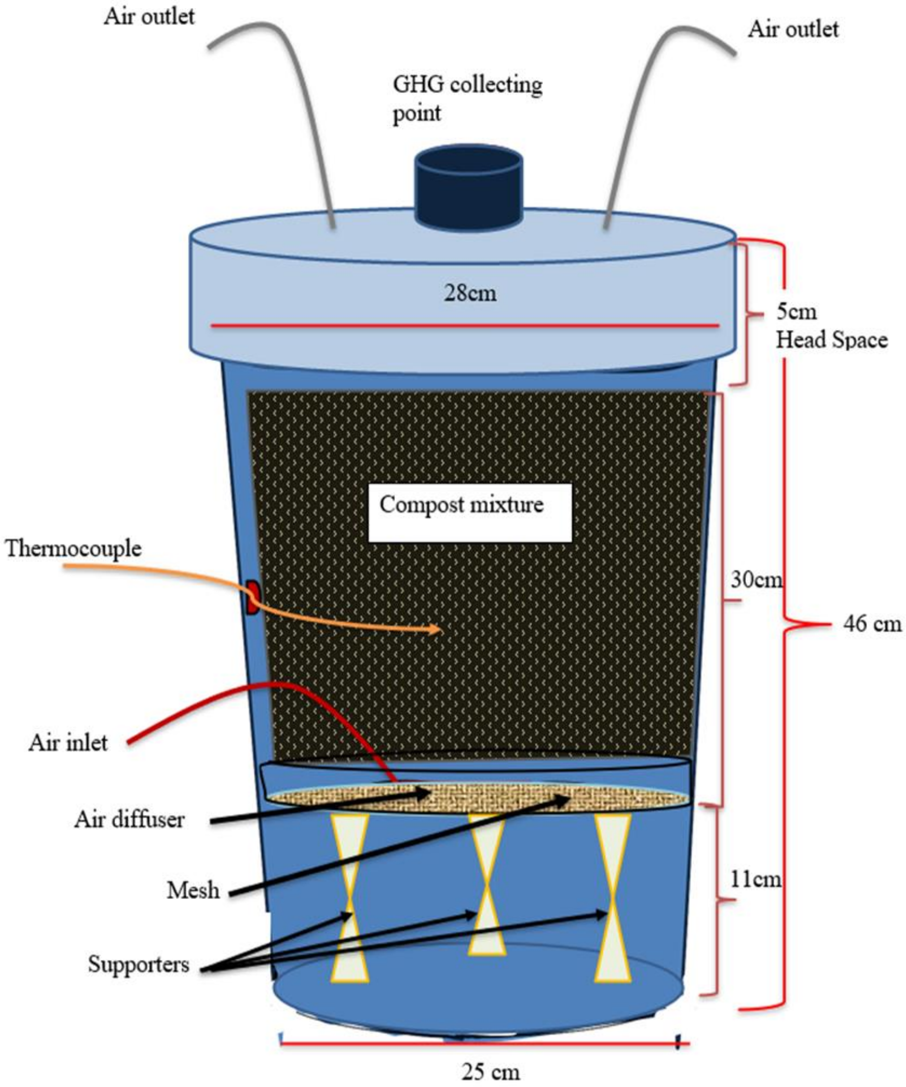

Figure 3.4 Vertical section of the compost bin.

The aeration rate was 4 L/min. Pumps were connected to timers which were programmed for continuous aeration throughout the day, with a 30 min break after 15 min of aeration. At the bottom, 11 cm of space was left for more air circulation and to drain off excess water. In the lid, there were two vial outlets to allow for air exit. Before collecting GHG, these vials were closed using a luer lock. At the center of the lid, a rubber septum was installed to collect GHG manually. Headspace was maintained at 6 L level in all the bins. Compost bins were covered with insulating materials to prevent temperature loss during the winter.

3.3.2.2 Experimental design

Based on the results from the incubation study, the best treatments were selected for further experimentation. Treatments with WA were removed since the pH of the compost was high and because of its heavy metal contamination. T3 and T4 had less PS to WS ratio, while T1 and T2 had a high PS ratio to WS. This was the main difference between these two treatments, and T2 and T4 were BC-added treatments of T1 and T3, respectively, to check the effect on GHG emission and compost quality.

The final treatments (three times replicated) included:

1. T1 – PS: WS: PM → 6:2:1
2. T2 – PS: WS: PM: BC → 30:10:5:2
3. T3 – PS: WS: PM → 1:2:1
4. T4 – PS: WS: PM: BC → 5:10:5:1

C/N of all the treatments maintained at the same level at the initial stage (40±2:1). When calculating, the C/N of T2 and T4 C content of BC was not considered. The moisture of the compost mixtures was maintained at a 60% (wet weight basis). Samples were collected for all analyses on days 1, 14, 45, and 90. Laboratory analyses were carried out at Boreal Ecosystem Research Facility (BERF), GC-MUN.

3.4 Laboratory analysis

3.4.1 Total C and N

Samples were dried at 40°C and pulverized using a cryo mill. Then 2±0.5 mg was weighed into small tin capsules. Analysis was done by using Perkin Elmer Series II CHNS/O Analyser 2400. Analyzer use the combustion method to convert sample elements into simple gases, CO_2, H_2O, and N_2. Due to the malfunctioning of the CHN analyzer at BERF, GC-MUN, and to cross-check the values obtained by analysis at each sampling time, 10 g of wet samples were sent to the provincial laboratory for analysis (NLSPF).

3.4.2 Mineral nitrogen

Ammonium (NH_4^+) and nitrate (NO_3^-) were analyzed on a 1:10 compost to 2 M KCl (potassium chloride) solution ratio, as suggested by Maynard et al. (2007). Four grams of wet compost samples were weighed into 125 mL conical flasks, and 40 mL of 2 M KCl was added and shaken end-to-end for 45 min on a benchtop open-air platform shaker (Thermo Fisher Scientific, USA). After that, extracts were allowed to settle for another 45 min, filtered using 42 ashless filter paper to 20 mL scintillation vials and stored at -20°C until used for analysis. NH_4^+ and NO_3^- were analyzed using a flow inject analyzer (Lachat Instruments, Colorado, USA) at BERF.

3.4.3 pH and electrical conductivity (EC)

The method mentioned by Jørgensen (2009) was used for determining compost pH and EC. pH was measured by using OAKTON pH 700 benchtop pH meter in 1:5 (WW) wet solids to de-ionized (DI) water ratio. The same solution was used to measure EC using a benchtop OAKTON EC meter. The solutions were shaken end to end for 30 min on a benchtop open-air platform shaker (Thermo Fisher Scientific, USA) and then allowed to settle for 1 hour before measuring the pH and EC values.

3.4.4 Temperature

Using the Digi-key TP 29-ND thermocouple probe, the temperature was measured every day for the first two weeks and then once every three days for 90 days in both incubation and compost bin studies. The thermocouple was connected to a Digi-key dual input thermometer (BK710-ND, Digi-Key electronics, Canada) to get the readings.

3.4.5 Compost analysis

Compost analysis for soluble salts, total N, total C, and nutrient analysis for P, K, Ca, Mg, Fe, Mn, Cu, Zn, B, and Na was done at the provincial lab in St. John’s (NLSPF) using Inductively Coupled Plasma – Optical Emission Spectrometry (ICP-OES). Twenty grams of wet samples were sent to the lab for analysis collected at the 1st, 14th, 45th, and 90th day of both incubation and compost bin studies. This analysis was also done on raw materials before starting the experiment.

3.4.6 Dry matter content (DM)

The method described by Jørgensen (2009) was used to determine compost DM and Ash content. DM content was calculated by drying samples at 80°C until samples reached a constant weight and calculated as a percentage of wet weight. Then, dried samples were incinerated at 550°C for three hours in the muffle furnace to measure the ash content.

3.4.7 Germination test for compost maturity

This test was adopted by Luo et al. (2018). A commonly used seed variety for phytotoxicity tests was chosen for the test: Rudolph organic radish (*Raphanus sativus L*). This test has three steps: i) Preparation of compost extract, ii) incubating seeds with extract, and iii) measuring and calculating values (number of germinated seeds and total radical length).

i) Compost extraction preparation – 10 g of wet compost was measured in a 125 mL conical flask, and 100 mL of DI water was added. The mixture was shaken for 30 min at 150 rpm and left for 72 h to settle under covering. Then the solutions were filtered using 42 ashless filter papers (Guo et al., 2012).

ii) Seed incubation – 10 cm diameter disposable Petri dishes were used for the test. Twenty radish seeds were placed in between filter papers, and 10 mL of compost extraction was added. The lids were kept sealed and left in the dark for four days at room temperature (Luo et al., 2018). Eight dishes of control were made with 10 mL of DI water per dish.

iii) Measurements and calculations – After four days, the number of germinated seeds and radical length were measured. The germination index (GI) was calculated using the formula in the Equation 3.1 given by Luo et al. (2018).

After six days of germination, the seedlings were removed from the dish, and the length of the seedling was measured. These values were used to calculate the seedling vigor index (SVI) as shown by the Equation 3.2 (Ali et al., 2020).

$$Relative\ seed\ germination\ (RSG) = \frac{Number\ of\ germinated\ seed (sample)}{Number\ of\ germinated\ seeds (control)}$$

$$Relative\ radical\ growth\ (RRG) = \frac{Total\ radical\ length\ of\ germinated\ seeds\ (sample)}{Total\ radical\ length\ of\ germinated\ seeds\ (control)}$$

$$GI = RSG \times RRG \times 100\%$$

Equation 3.1 Formula used for germination index (GI) calculation (Luo et al., 2018).

$$Germination\ \% = \frac{Number\ of\ germinated\ seeds}{Total\ number\ of\ seeds} \times 100\%$$

$$SVI = \frac{Seedling\ length\ (cm)}{100} \times Germination\ \%$$

Equation 3.2 Formula used for seedling vigor index (SVI) calculation (Ali et al., 2020).

3.4.8 Greenhouse gas (GHG) collection and analysis

3.4.8.1 Incubation study

GHG samples were measured using a Gasmet DX 4015 instrument daily during the first week, at three-day intervals until the third week, and then weekly until the experiment ended nine weeks later. As shown in Figure 3.5, Gasmet DX 4015 was directly connected to Calcmet software; it gives real-time results on GHG and ammonia (NH_3) concentration results, sample and reference spectra, and time trends for the analyzed gases. Before collecting samples, lids with sample vials were secured into jars and left for five minutes to concentrate gases inside the jar.

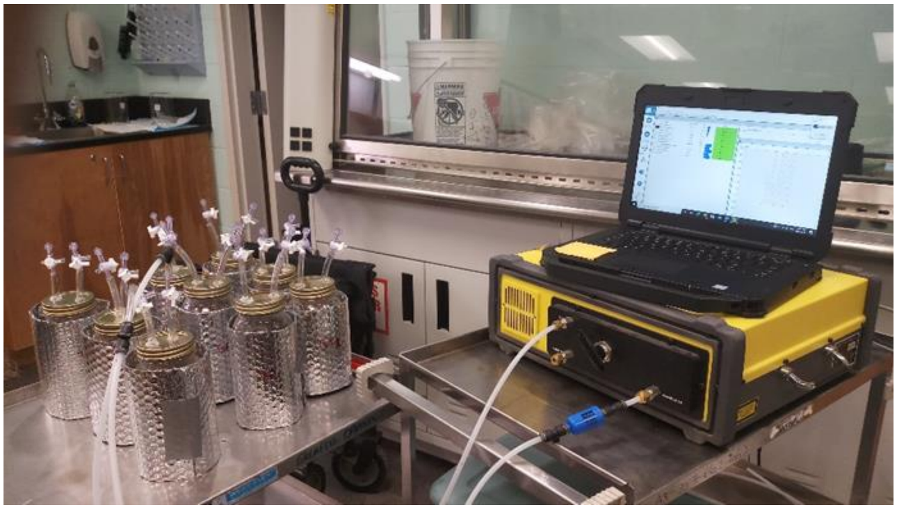

Figure 3.5 GHG sample collection and analysis during incubation study.

3.4.8.2 Compost bin

GHG samples were manually collected using a syringe into 15 mL EPA vials, as shown in Figure 3.6, daily during the first week and then every three days until the third week. Afterward, samples were collected weekly for twelve weeks. Thirty minutes before collecting samples, aerators were switched off, and air outlets were sealed using luer locks. Sample vials were sent to the GHG analysis lab at Dalhousie University Agricultural Campus, Truro, NS, Canada.

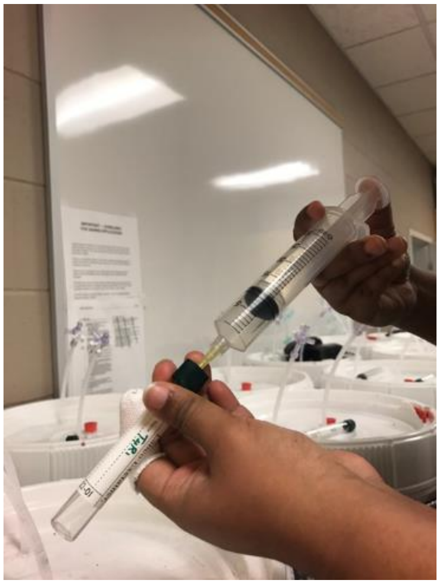

Figure 3.6 This picture shows how GHG samples were manually collected.

3.4.9 Statistical analysis

One-way ANOVA with Fisher's pairwise comparison on Minitab Statistical Software was used to evaluate the significant differences among treatments for GHG and NH_3 emissions, temperature, pH, EC, NO_3^-, NH_4^+, moisture %, total C, total N, C/N, and GI variability in both the compost bin and incubation study.

A repeated measures ANOVA with sphericity assumed ($P > 0.05$) and post-hoc test with Bonferroni correction for difference over time was performed on XLSTAT software (Premium, 2022) for total N, total C, and other nutrients in compost bin.

CHAPTER 4: Results and Discussion

4.1 Incubation study

T1 – PS: WS: PM → 6:10:6, T2 – PS: WS: PM: BC → 6:10:6:0.5, T3 – PS: WS: PM → 11:6:4, T4 – PS: WS: PM: BC → 11:6:4:0.5, T5 – PS: WS: PM: WA → 6:6:3:1, and T6 – PS: WS: PM: WA: BC → 6:6:3:1:0.5 were the treatments used in this study. (PS–Paper sludge, WS–Wood shavings, PM–Poultry manure, WA–Wood ash, BC–Biochar)

4.1.1 Temperature

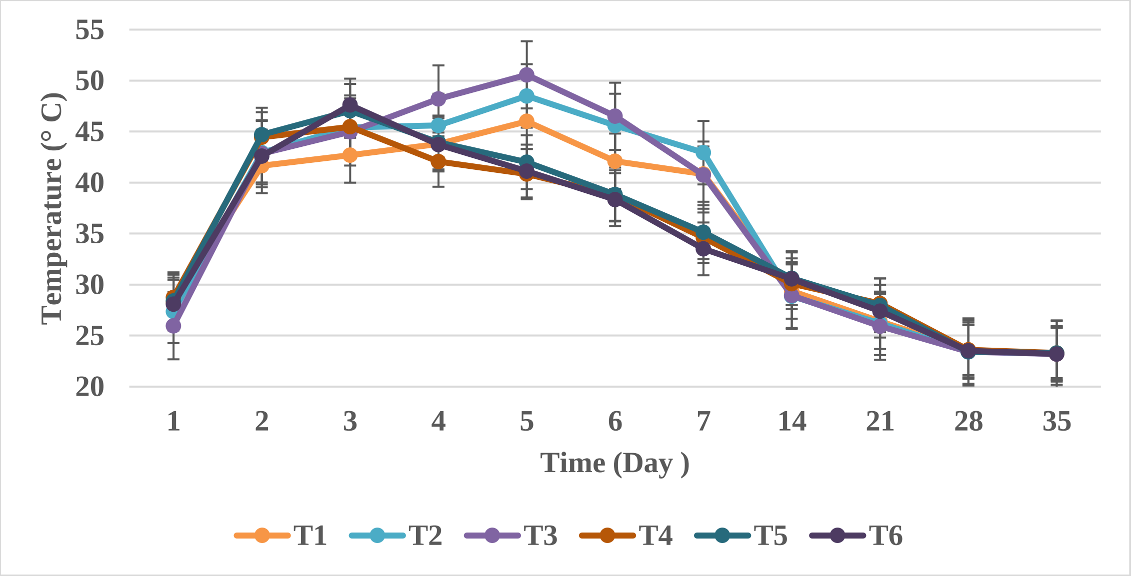

Figure 4.1 The temperature variation in compost bins during first 35 days. After day 28 to day 60 temperature was at room temperature level. Error bars represent standard error.

During the first week of the incubation study, the temperature of T1, T2, and T3 showed the highest values on day five, while T4, T5, and T6 showed the highest peak on day three (Figure 4.1). The highest average temperature of 50.6°C (SD = 10.9) was reported in T3. After days three and five, the temperature fell gradually and reached room temperature on day 28 where it stayed for the rest of the time period. The thermophilic phase was found to be from day 2 to 7. From day 5 to day 7, one-way ANOVA shows that there was a significant difference ($P < 0.001$) between T1, T2, and T3 to T4, T5, and T6. According to Fisher pairwise comparison, T1, T2, and T3 have high mean values, belonging to the same mean group. Based on the results of the two sample t-test, there was no significant difference among the mean values.

An increase in temperature during the initial stage of the composting process is crucial for OM degradation (Chan et al., 2016), which indicates aerobic microbial decomposition of OM (Charest et al., 2002). This temperature profile matches the temperature profile of other studies (Chowdhury et al., 20013; Chan et al., 2016; Charest et al., 2002) and the temperature profile of the compost bin experiment done later.

4.1.2 pH

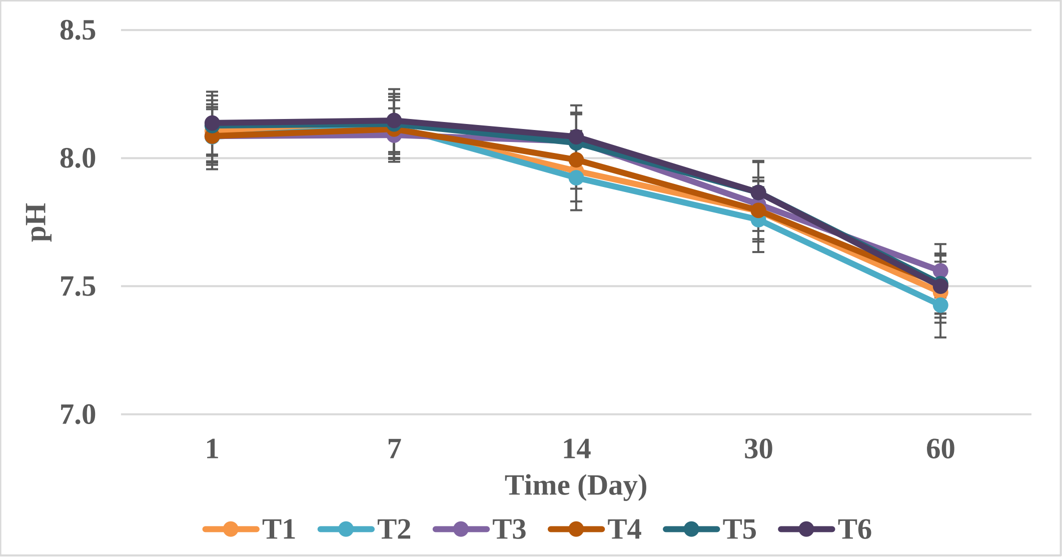

Figure 4.2 pH changes over time of each treatment. Error bars represent standard error.

The temporal variability of pH shows a slight increase in the first seven days, during the thermophilic stage but the measurable difference would not be considered significant (Figure 4.2). From T1 to T6, pH increased by 0.16,0.49, 0.04, 0.32, 0.08, and 0.12 %, respectively. After day seven, pH decreased gradually, showing 7.77, 8.12, 6.51, 7.17, 7.59, and 7.82% decline from initial pH values from treatment 1 to 6, respectively. This pH decrease may be due to microorganisms' release of H^+ during the nitrification process (Liu et al., 2019). Fluctuation of pH was observed during the thermophilic stage due to the decomposition and fixation of organic C and N (Guo et al., 2012). According to the one-way ANOVA, there was no significant difference among treatments for pH over time (P = 1).

T5 and T6 have WA, which was added to check the WA effect on the pH of compost, but it did not show any significant effect during the whole composting period. During the entire time, average pH values varied between 8.1 and 7.5, which was favorable for the composting process (Guo et al., 2012).

4.1.3 Electric Conductivity (EC)

According to the one-way ANOVA, there was no significant effect among six treatments for EC variation over time (P= 0.996). As shown in Figure 4.3, EC increases over time in all treatments, of which 61.32, 67.64, 61.51, 53.75, 62.98, and 57.34 % increments were shown from T1 to T6 over time. These values are similar to the observed data in the PS composting experiment done by Tripepi et al. (1996), and Charest and Beauchamp (2002), but EC values are higher than the recommended value of 0.7–2.0 dS/m by A & L Laboratories (2004).

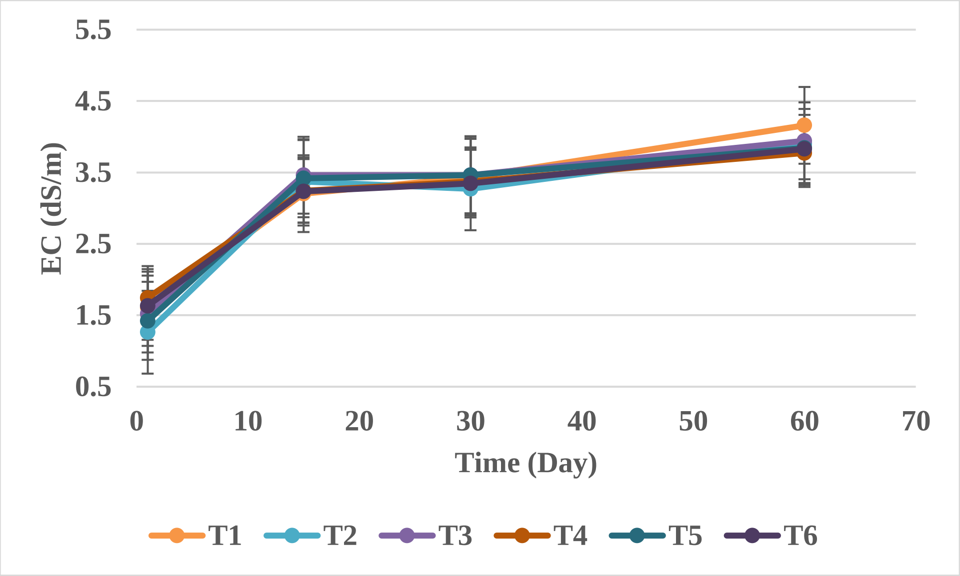

Figure 4.3 Electrical conductivity (EC) changes over time of each treatment. Error bars represent standard error.

4.1.4 C/N

Initial C/N is crucial for controlling NH_3 loss during composting (Sikora, 1999; Larsen and McCartney, 2000). Their results show that an initial C/N increase to 30 or above reduces NH_3 loss at the initial composting stage.

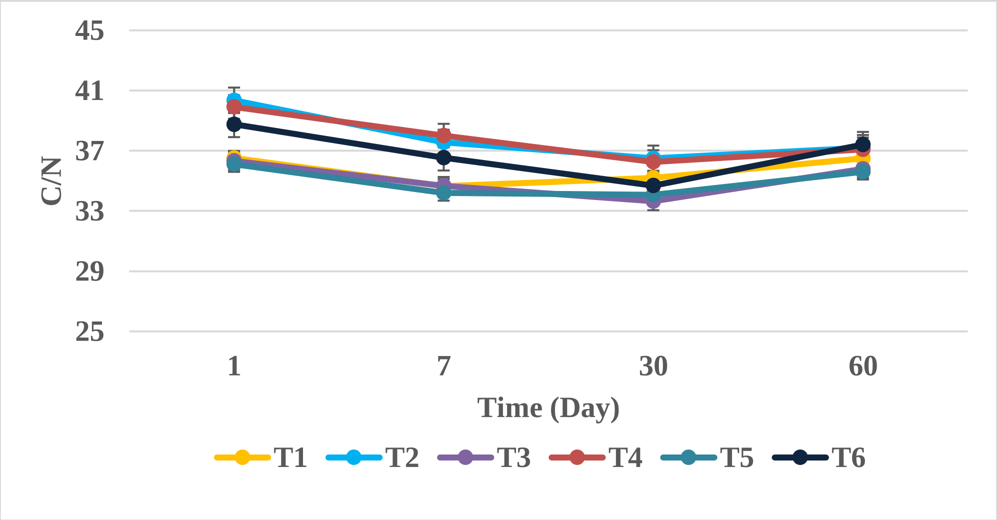

Figure 4.4 Carbon nitrogen ratio (C/N) changes over time of each treatment. Error bars represent standard error.

According to one-way ANOVA, there is no significant difference between (P= 1) treatments for C/N variation over time. The first-day experiment was set up to maintain the same range of C/N for all the treatments (38±2), and after 60 days, C/N decreased in all treatments (Figure 4.4). Total C/N loss values of 0.050, 7.81, 1.44, 7.040, 1.40, and 3.50 % were shown in T1 to T6, respectively. The mean of the final C/N of all treatments lies above 35, which shows a lack of OM degradation, which is also an indicator of immature compost. According to Canadian compost standards (BNQ, 1997), none of the products from different treatments should be considered matured compost.

4.1.5 Inorganic nitrogen

As shown in Figure 4.5, the concentration of NH_4^+–N is high at the initial days of composting, rapidly decreases during the first 6 days, and then gradually decreases over time and stabilizes after 30 days. This decrease can be described by the increase in NH_3 volatilization after the first week of compost (López-Cano et al., 2016). During NH_3 volatilization, N losses happen (Charest and Beauchamp, 2002). One-way ANOVA shows that there were no significant differences (P= 0.995) among treatments for NH_4^+–N concentration variation over time.

The concentration of NO_3^-–N was low during the first seven days of composting, and then gradually begins to increase. During the thermophilic phase, nitrification does not occur due to the inhibiting effect of high temperature and the excessive amount of NH_4^+–N (Li et al., 2019). When temperature decreased, NO_3^-–N increased and reached its maximum concentrations of 66.1, 67.0, 67.77, 68.41, 63.6, and 69.70 mg N/L in treatment from T1 to T6, respectively. According to one-way ANOVA, there is no significant difference (P = 1) among treatments for the variability of NO_3^-–N concentration.

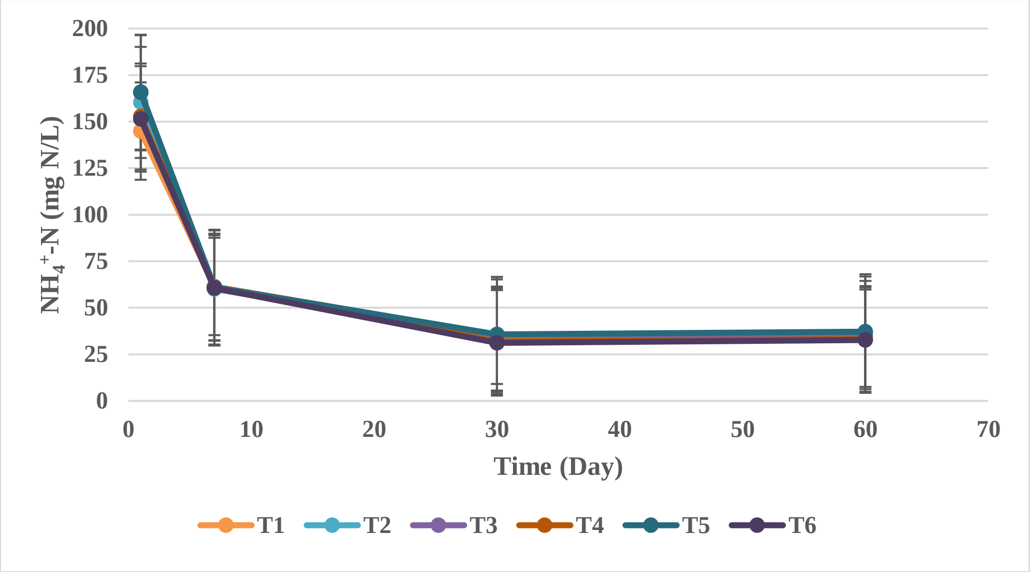

Figure 4.5 Evolution of ammonium (NH_4^+–N) with time. Error bars represent standard error.

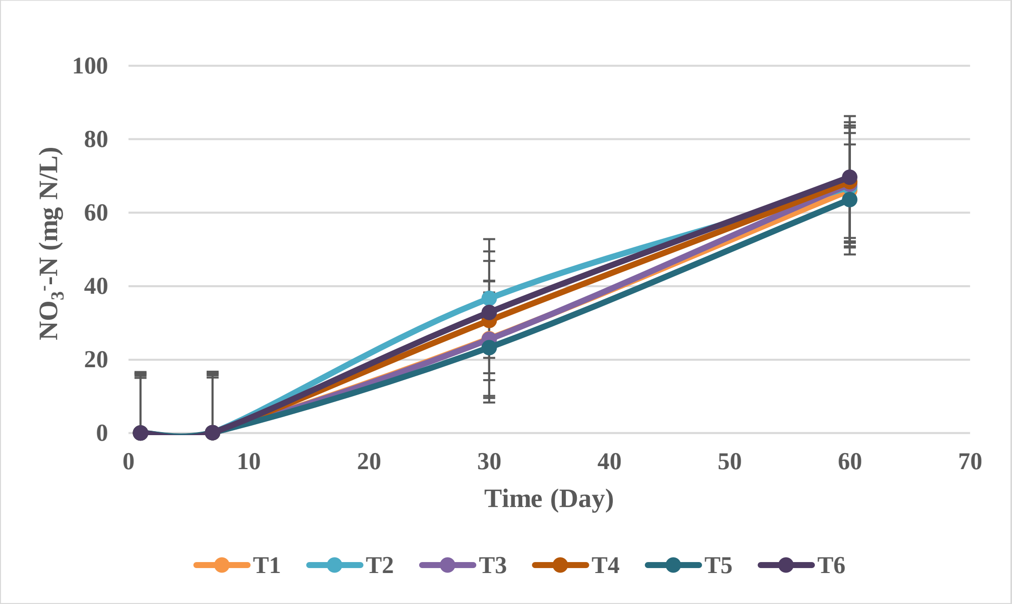

Figure 4.6 Evolution of nitrate (NO_3^-– N) with time. Error bars represent standard error.

4.1.6.1 Ammonia and nitrous dioxide emission

At the beginning of the thermophilic stage, NH_3 emissions were low but gradually increased and continued until day 60. An increase in NH_3 emission causes the loss of organic N, which may have resulted in the increase of the C/N (Figure 4.4). The highest NH_3 peak was observed on day 28 for all the treatments, which was similar to the day 60 profile (Figure 4.7). One-way ANOVA showed that there were no significant differences (P = 1) among treatments for NH_3 emissions during the 60 days.

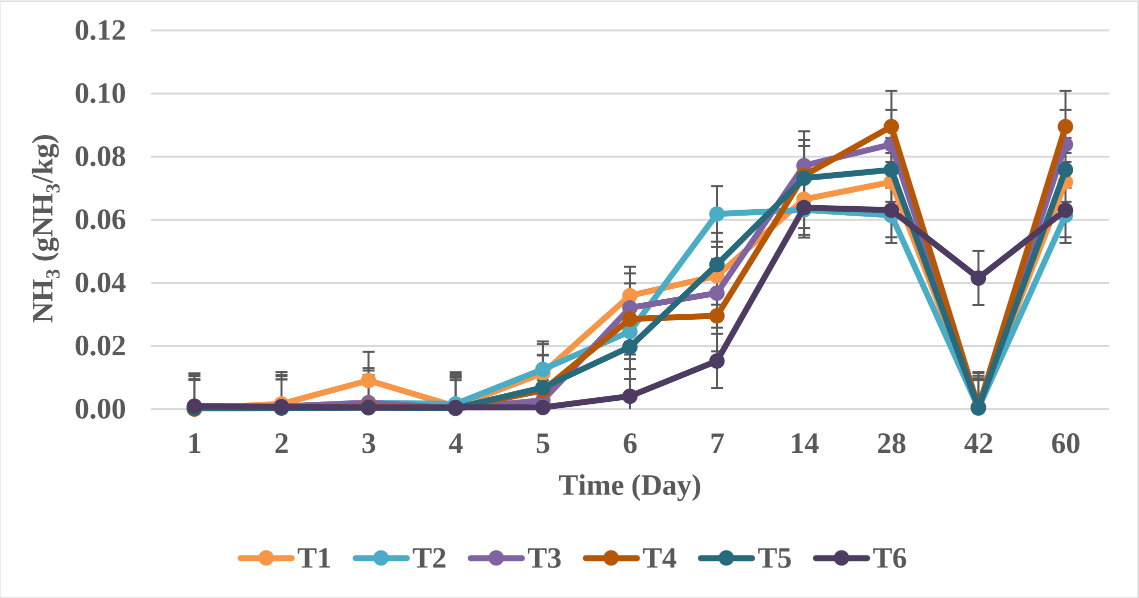

Figure 4.7 Ammonia emission (NH_3) with time. Error bars represent standard error.

N_2O emissions happen under both aerobic and anaerobic conditions (Chowdhury et al., 2014). Normally N_2O is produced by the nitrification of ammonium nitrogen and denitrification of nitrate nitrogen (Chen et al., 2018). As shown in Figure 4.8, in the incubation study during the thermophilic stage, no peak for N_2O emission was observed. Similar results were observed in other studies done by several authors (Chowdhury et al., 2014; Chen et al., 2018; Chen et al., 2020). This may be due to a lack of available N in the mixtures as the quantity of material in the jars was very small (around 250 g of the mixture on average in 500 mL jars). During the whole time period, the concentration of N_2O varied between 275 and 375 mg/kg for all five treatments except for T1. T1 shows the highest peak emission of N_2O emission on day one and another peak emission on day 14. According to one-way ANOVA, there was no significant difference among treatments for N_2O emission ($P = 0.135$).

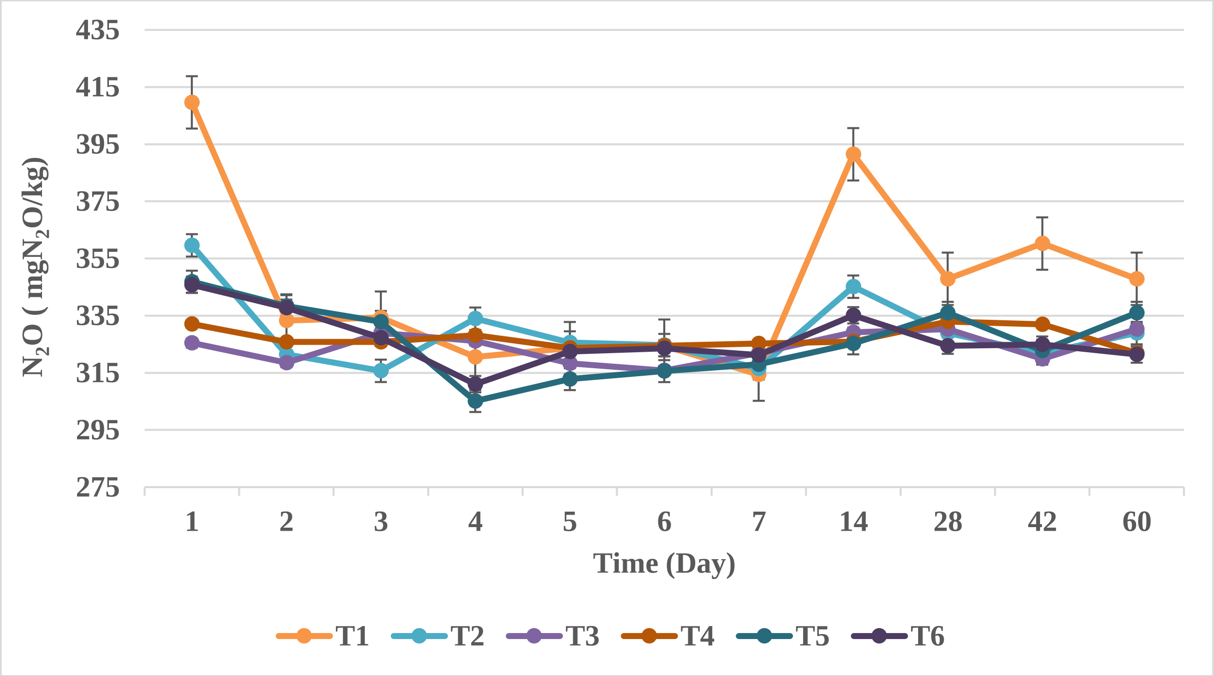

Figure 4.8 Nitrous dioxide (N_2O) emission with time. Error bars represent standard error.

4.1.6.2 Carbon dioxide and methane emissions

During the first week, CO_2 and CH_4 show high emissions. During the thermophilic stage, due to high microbial activities, CO_2 and CH_4 volatilization can be observed (Sun et al., 2016). This is similar to other experiments done by Chowdhury et al. (2013). Sun et al. (2014). and Sun et al. (2016) observed a similar pattern in their compost bin experiment.

According to one-way ANOVA and Fisher pairwise comparison, there is a significant difference ($P < 0.001$) between T5 and T6 to T1, T2, T3, and T4 during the first week. T5 shows the highest peak of CH_4 emission on day six out of all the treatments. T5 and T6 contain WA in their mixtures. This shows that there may be some effect of WA on triggering CH_4 emissions. The highest peak of CH_4 emission was observed during day 5 for T5, T4, T2, and T1, while treatments T6 and T3 showed the highest peak on day 4. Overall, the highest CH_4 emission was observed in the T5 at the concentration of 0.00311 g/kg (Figure 4.9)

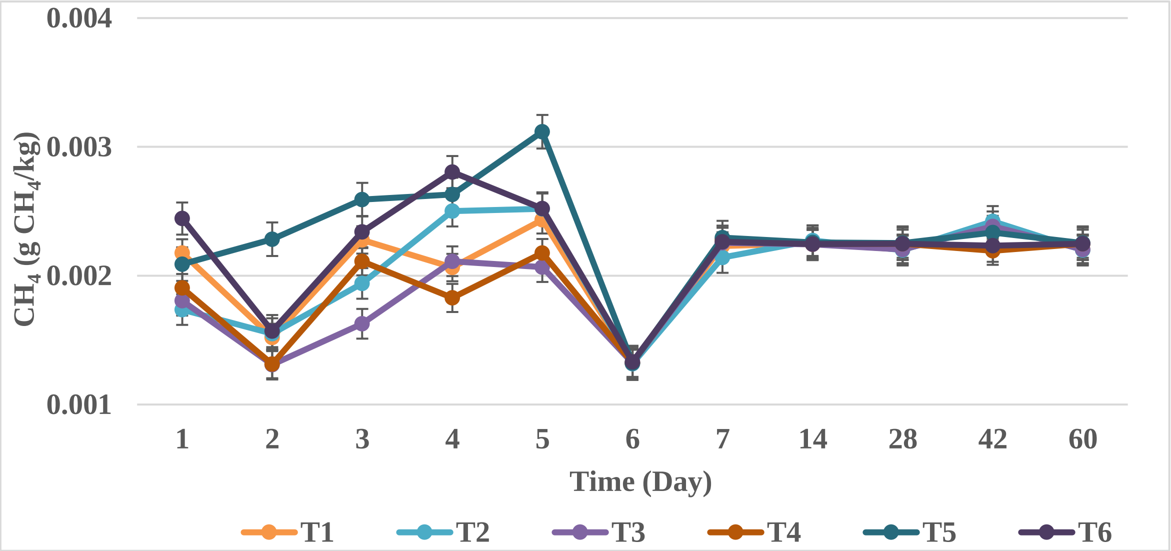

Figure 4.9 Methane (CH_4) emission with time. Error bars represent standard error.

According to one-way ANOVA ($P = 0.922$), there were no significant differences found in the emission of CO_2 during the length of the experiment. Day 1 shows the highest emission in all the treatments for CO_2 (Figure 4.10), similar to the N_2O profile. After the recorded peak on day 1, CO_2 emission gradually decreases from day 3 to 60 with minor fluctuations; these fluctuations are similar to other results shown in studies by Chowdhury et al. (2013), Sun et al. (2014) and Sun et al. (2016).

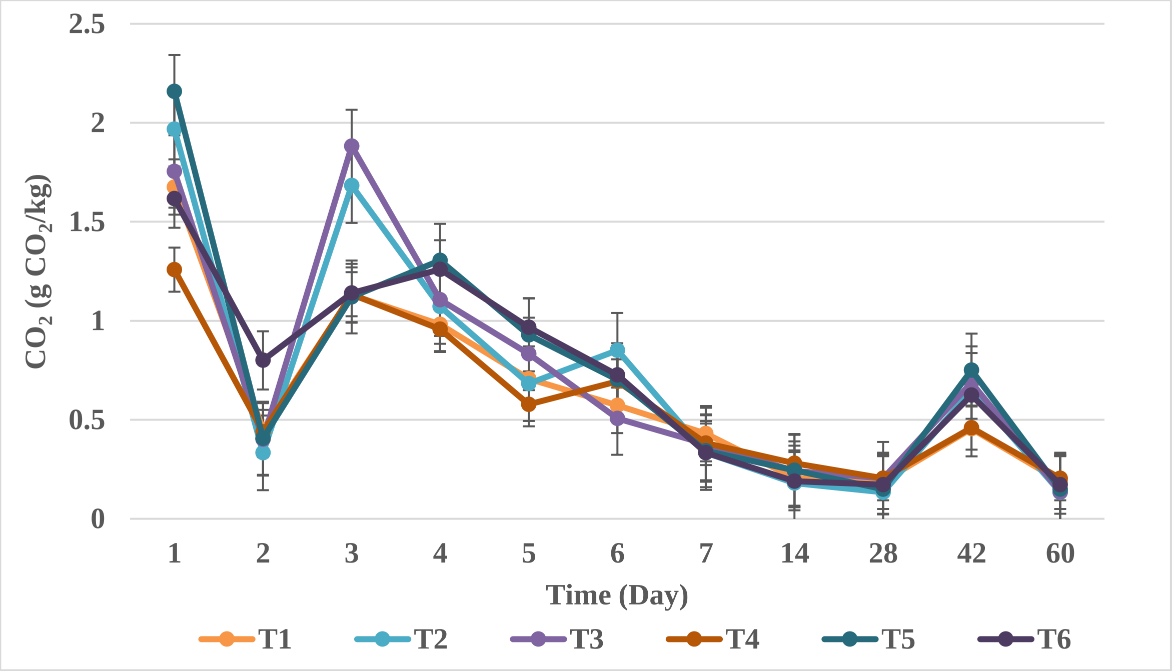

Figure 4.10 Carbon dioxide (CO_2) emission with time. Error bars represent standard error.

Overall, the addition of BC to T2, T4 & T6 did not show any significant effect on GHG or NH_3 emissions during this incubation study. Also, WA did not show any impact on the pH of the mixtures. Therefore, during the compost bin experiment, T5 and T6 were omitted. As the recorded temperature was higher in the current incubation study than in previous trials, it was decided to use the same mixing combinations with the same C/N for the continuing compost bin experiment.

4.2 Compost big bins

Four treatments used during the experiment were as follows.

1. T1 – PS: WS: PM → 6:2:1
2. T2 – PS: WS: PM: BC → 30:10:5:2
3. T3 – PS: WS: PM → 1:2:1
4. T4 – PS: WS: PM: BC → 5:10:5:1

4.2.1 Temperature

In the compost bins during days from 2 to 4, the temperature reached the highest in all treatments. The highest temperature (50.1°C) was recorded on day 3, with an average of 44.8°C (SD = 9.51) for all the bins. T1 had the highest temperature during the experimentation period. According to one-way ANOVA, there was a significantly (P = 0.01) higher temperature in T1 and T2 compared to T3 and T4 during the first five days of the experiment. After the first week, there was no significant difference in the temperatures among all treatments. After day 49, the temperature reached room temperature in all the bins.

Temperature is a critical factor in the composting process. It is a physical indicator showing organic matter degradation by microbial activities (Chen et al., 2018). It is essential to reach a compost temperature above 55°C and maintain a minimum of 5 days to destroy the pathogenic microorganisms to indicate the hygiene of the final compost product (Yang et al., 2013). During the experiment, the temperature did not reach a high level and was not consistent. This is possibly due to a lack of readily available C for microbial digestion, as PS and WS are high in lignocellulose content and low in available C (Sun et al., 2014) and due to lack of material since the bin only holds around 3 kg of the mixture. High temperatures and small thermophilic phases are often observed in small-scale laboratory experiments (Chowdhury et al., 2014). This is due to the use of homogeneous compost material used in small-scale experiments rather than large-scale windrow composting, where compost is more heterogeneous. According to Lo et al. (1993), compost without forced aeration maintained longer thermophilic conditions than the aerated compost pile. Also, most empirical data indicates that temperatures around 45°C to 65°C are ideal for composting (Chowdhury et al., 2014). T1 and T2 showed temperatures above 45°C.

As in Figure 4.11, the composting process shows three clear phases (as mentioned in section 2.4 in this thesis): phase 1 from day 1 to 2, where temperature quickly rises (mesophilic phase I); phase 2, where the high temperature was maintained from day 2 to day 4 (thermophilic phase); and phase 3, where the temperature starts to gradually drop after day 4 (mesophilic phase II). This matches the temperature profiles of other studies (Charest et al., 2002; Chowdhury et al., 2014).

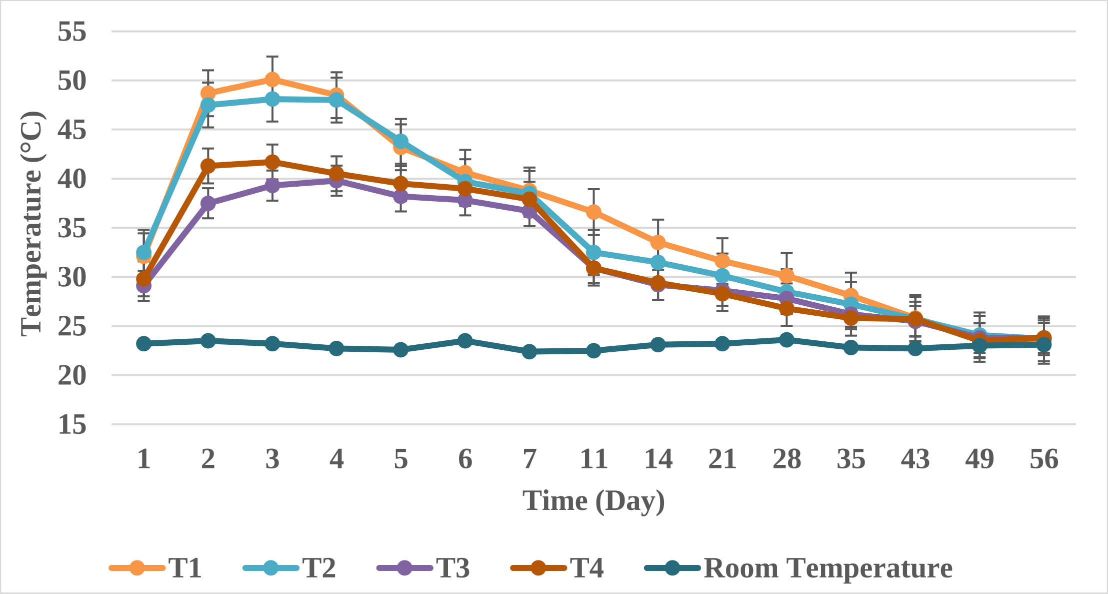

Figure 4.11 The temperature variation in compost bins during first 56 days. Between day 49 to day 90 temperature was at room temperature level. Error bars represent standard error.

4.2.2 Other physicochemical characteristics

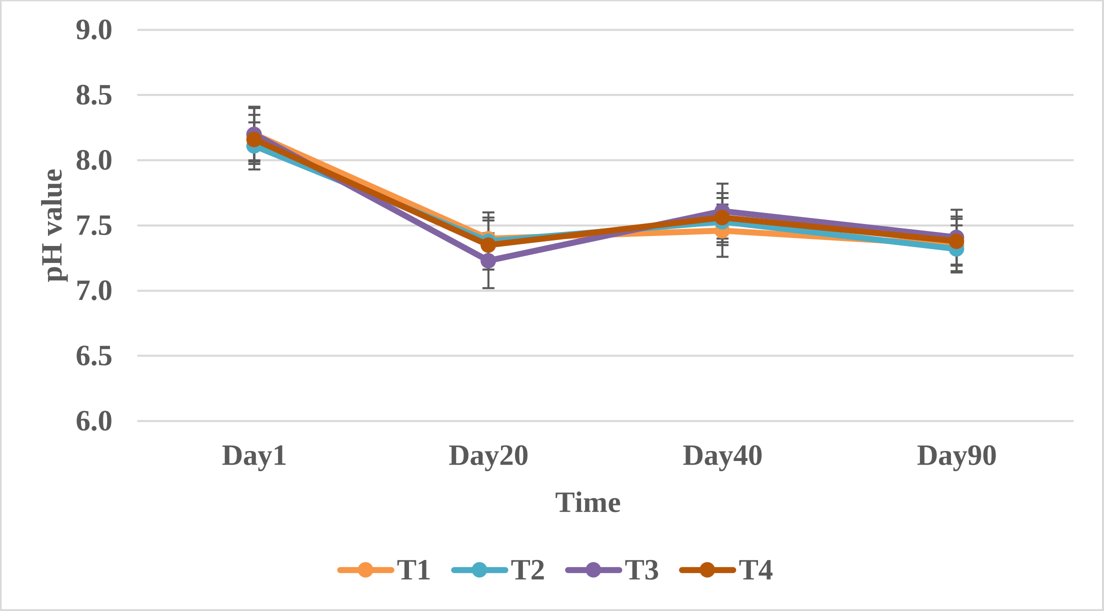

Figure 4.12 pH changes over time of each treatment. Error bars represent standard error.

As shown in Figure 4.12, there was no significant ($P = 1$) difference among treatments for the pH during the whole composting period, according to the one-way ANOVA. pH is one of the critical factors affecting GHG emissions and microbial activities. Also, Szanto et al. (2007), Wong et al. (2009), and Chan et al. (2016) suggested that a pH of 7.0–8.5 is needed for efficient composting. From day 1 to day 90, pH of all treatments decreased on average from 8.16 to 7.36. This pH reduction may be primarily due to the mineralization of P and N and the production of CO_2 and NO_3^- (Sun et al., 2014). From day 1 to day 14, pH decrease; this may be related to ammonia volatilization, production of organic acid, and CO_2 emission (Chen et al., 2018). After day 14, NO_3^- amount in all treatments had increased (Figure 4.14b). There is a high positive correlation (Pearson) between pH and NH_4^+ ($r = 0.802$, $P < 0.001$), and moderate negative correlation between pH and NO_3^- ($r = -0.609$, $P < 0.001$). With time ammonification decreased, and continuous OM degradation supposedly formed bicarbonates (Chen et al., 2018), stopping further reduction of pH, and the final pH of compost remained in the alkaline range. Overall, compost pH was in the range of 5.5 to 9.0, which is recommended by A&L Laboratories, 2004 for compost.

According to one-way ANOVA, there was no significant difference ($P = 0.952$) among the treatments for the EC during the whole composting period. With time, the EC of all treatments increased (Figure 4.13) by 61.0, 67.5, 64.4, and 54.17, 64.1% from T1 to T4 from day 1 to day 60. High EC leads to the phytotoxicity of end product compost (Sullivan et al., 2018).

Inorganic N is the total of NH_4^+−N and NO_3^-−N. In Figure 4.14, NH_4^+−N decreases with time in all the treatments, and according to one-way ANOVA, there was no significant difference ($P = 0.981$) among treatments. Microorganisms immobilize NH_4^+−N leading to a decrease of NH_4^+−N with time during the composting process. During the initial stages, a drastic drop in NH_4^+−N concentration reflects the organic N ammonification as OM degradation due to high microbial activities during the first week, as reflected by the higher temperatures.

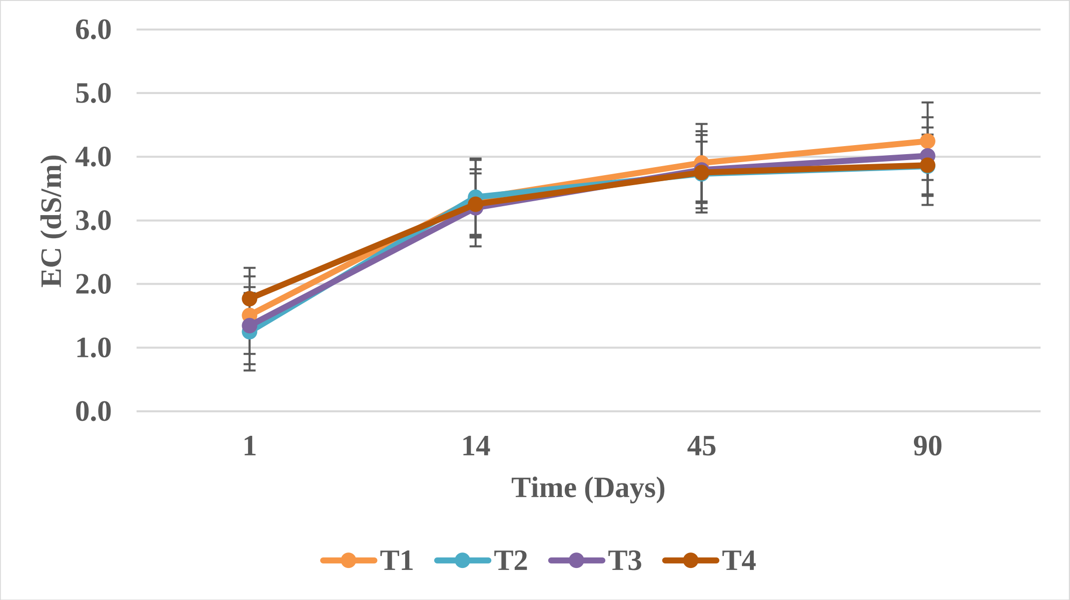

Figure 4.13 Electrical Conductivity (EC) changes with time during the compost experiment. Error bars represent standard error.

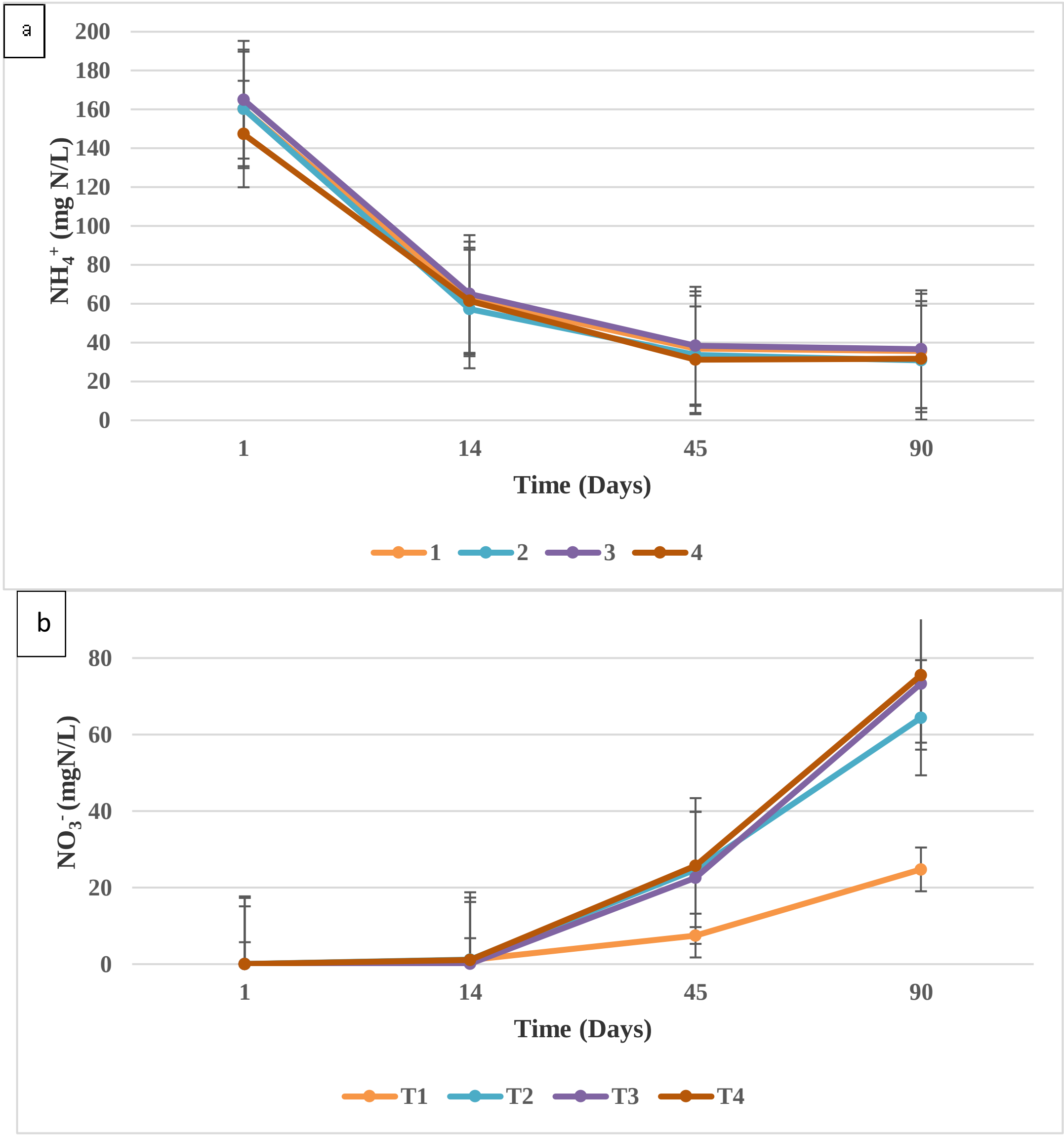

Figure 4.14 Evolution of ammonium (NH_4^+−N) and nitrate (NO_3^-−N) with time. Error bars represent standard error.

All treatments show an increase of NO_3^-−N during the maturation period of composting (from day 45 to day 90) when the temperature is close to ambient temperatures. An increase of NO_3^-−N shows the nitrification process happens in compost bins. The nitrification process in compost with materials like high PS is limited by low N availability as high lignocellulosic structure protects N. Previous studies have found that lignin is one of the plant polymers that control N mineralization during the decomposition of plant-derived by-products (Wang et al., 2018). Adding BC may improve nitrifying activities and enhance nitrification while making the environment suitable for nitrifying bacteria (Sánchez-García et al., 2015; López-Cano et al., 2016).

As shown in Figure 4.14b and supported by one-way ANOVA with Fishers' pairwise comparison T1 shows a significant difference with the rest of the treatments from day 45 to 90 (, $P < 0.001$). In T1, it has higher PS ratio than T3 and T4, and even though T2 has the same ratio for the materials as T1, it has the addition of BC, which increases the nitrification process in T2 more than in T1; this explains the difference of T1 from other treatments in having low NO_3^-.

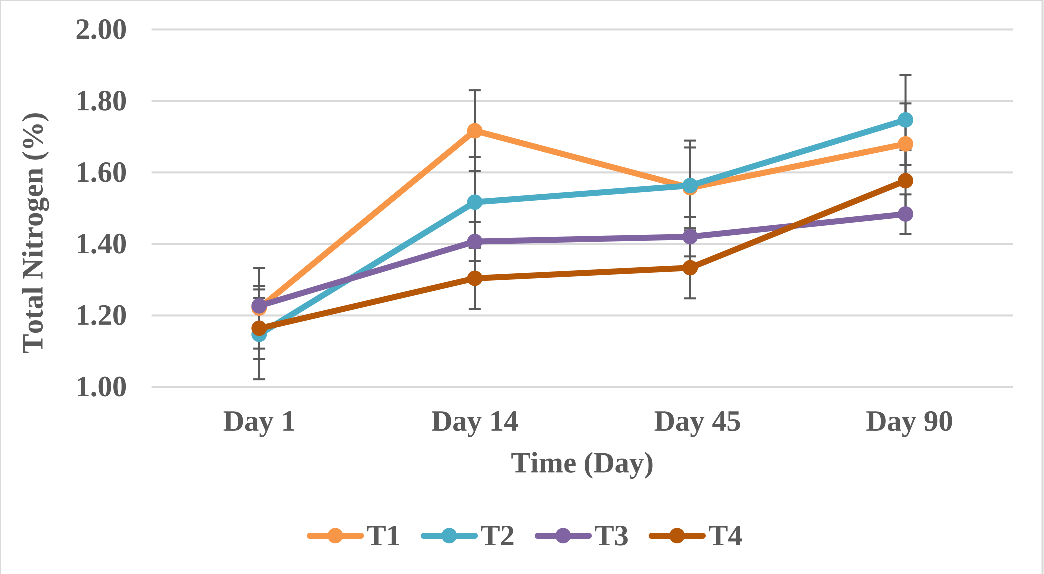

Figure 4.15 Mean values of total nitrogen increased over time. Error bars represent standard error.

According to repeated measures ANOVA and post- hoc test with a Bonferroni correction, there was a significant difference in total N means over time ($P < 0.001$) T1, T2, T3, and T4 increase total N from day 1 to day 90 by 27.38, 34.35, 17.30, and 26.21%, respectively. Many other compost experiments also have observed an increase in total N over time (Charles et al., 2002; Thambirajah et al., 1995). This happens as a result of C biodegraded in PS and concentrated in compost piles and mainly unconstrained through CO_2 (Charles and Beauchamp, 2002).

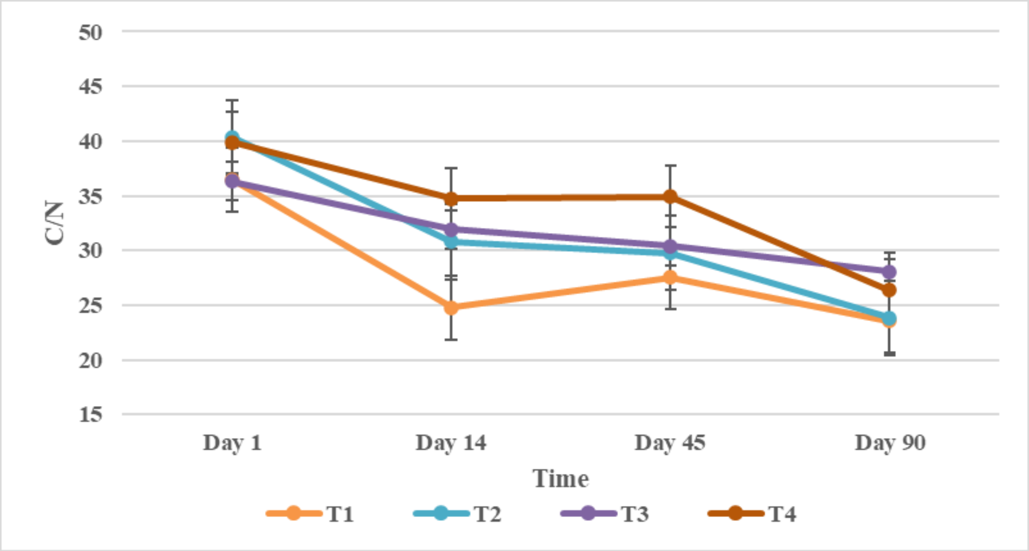

Figure 4.16 Mean values of C/N changes over time. Error bars represent standard error.

The initial C/N of all treatments was maintained at the same level. The time C/N of T1 to T4 decreased by 35.43, 40.95, 22.68, and 33.98 %, respectively. According to one-way ANOVA Fisher pairwise comparison on day 14, there was a significant difference among treatments ($P < 0.001$). T1 (mean = 3.209) significantly differs from T4 (mean = 3.54), T3, and T2 (mean = 3.46). T3 and T2 share the same mean group. On day 45, there is a significant difference between T4 and the rest of the treatments ($P = 0.008$). Literature has also reported that the initial C/N is a key factor in controlling ammonia losses during the composting process (Sikora and Sowers, 1985; Ekinci et al., 2000; Larsen and McCarthney, 2000), according to Charles and Beauchamp (2002), N is mostly lost through ammonia volatilization. Even though the mean C/N during 90 days of the experiment was reduced (mean values T1 and T2 (24:1), T3 (28:1), and T4 (26:1)), according to

Canadian compost standards (A & L Laboratories, 2004), none of the compost treatments can be considered as matured compost since their C/N are above 15:1 which is not suitable for agriculture purpose. Lack of temperature increase, or short thermophilic phase, size of the trial, or lack of microbial activities could be the reason for this.

Nutrient analysis is a way to determine the suitability of compost as a soil amendment. Each nutrient has an optimal level where plants grow best. The data presented below indicates which nutrients are within the optimal range. A repeated measure ANOVA was performed for all nutrients over time, and post-hoc test with Bonferroni correction was used to detect the difference over time.

Phosphorus (P) and potassium (K) are essential for plant growth (Appiah-Effah et al., 2016). According to Appiah-Effah et al. (2016), P deficiency is the second most important soil fertility problem throughout the world, and the excessive amount of P can immobilize other chemical elements such as zinc (Zn) and copper (Cu). P variation during the composting process is shown in Figure 4.17. From day 1 to 90 the overall mean of P concentration increased by 17.9, 20.8, 20.8, and 25.6% from T1 to T4, respectively. This rise could be a result of the microbial decomposition of OM. There was a significant difference between T1 and T4 for P concentration variation: Day 1 ($P = 0.07$) and Day 90 ($P = 0.002$) and over time ($P < 0.001$). Similar variations in P percentage were observed in studies done by Shweta et al. (2010) and Byakodi and Babu (2022).

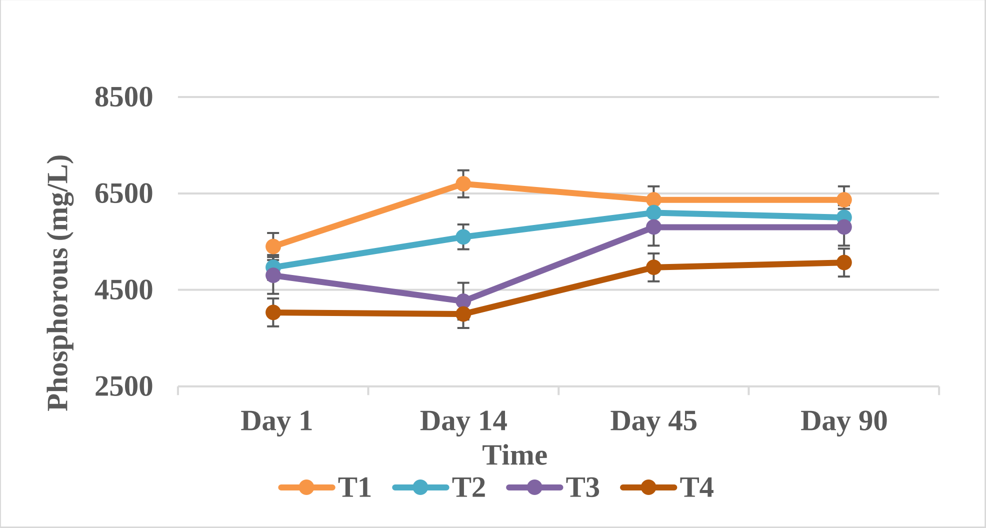

Figure 4.17 Mean variation of phosphorus (P) concentration over time. Error bars represent standard error.

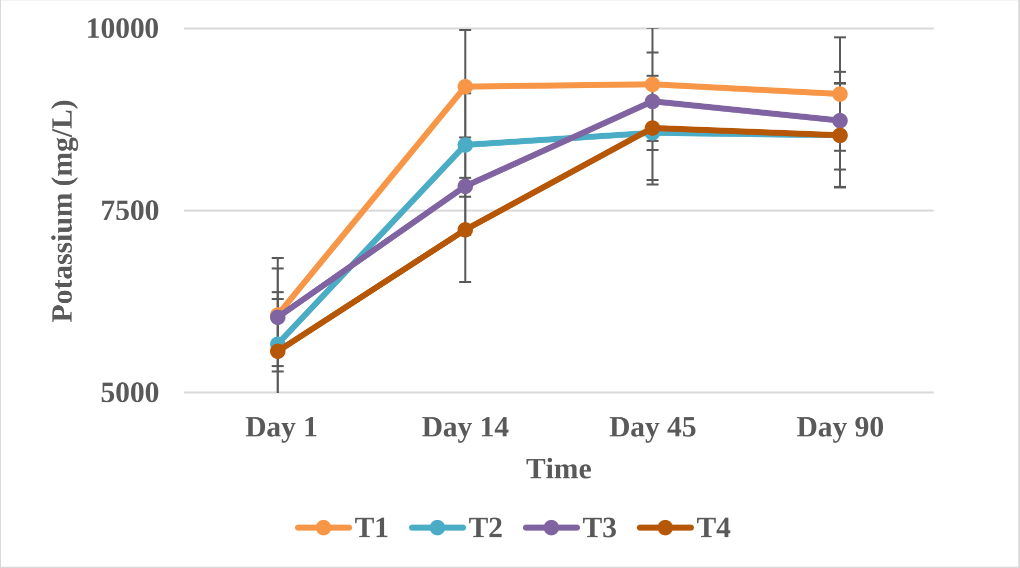

Figure 4.18 Mean variation of potassium (K) concentration over time. Error bars represent standard error.

There was a significant difference in K over time ($P < 0.001$). As shown in Figure 4.18, from day 1 to 90, the mean K concentration increased by 50.0, 50.5, 44.74, and 53.2% from T1 to T4, respectively. The increase in K concentration at the later part of the composting process is attributed to potassium fixing during the mesophilic phase of composting (Byakodi and Babu, 2022). Increased microbial activity increases the mineralization rate and acid production that solubilizes K making it more readily available in compost (Gupta and Garg, 2008). K is an important plant macronutrient used in amino acids and protein synthesis and it helps to regulate water flow inside plants (Appiah-Effah et al., 2016).

There was a significant difference between days 1 and 14 for Zn concentration increment (P = 0.003). And there was a significant difference between treatments on day 14. T1, T2, are significantly different from T3 and T4 (P < 0.001). As shown in Figure 4.19, T2 and T4 have lower Zn concentrations than T1 and T3, respectively, at the end of the composting. This may be due to the adsorption of Zn by BC (Hua et al., 2009).

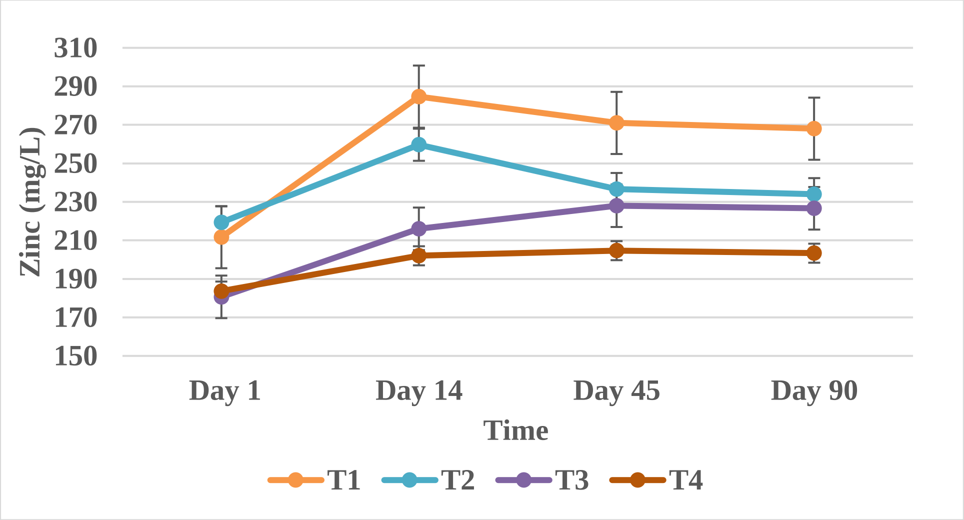

Figure 4.19 Mean variation of zinc (Zn) concentration over time. Error bars represent standard error.

There was no significant difference in the means of Cu over time (P = 0.572). After 90 days T3 and T4 showed a 1.17% and 3.70% increase in the mean concentration, while T1 and T2 showed a 0.94% and 4.39% decrease in the mean concentration of Cu. As shown in Figure 4.20, T1 and T2 showed higher mean values for Cu concentration over time.

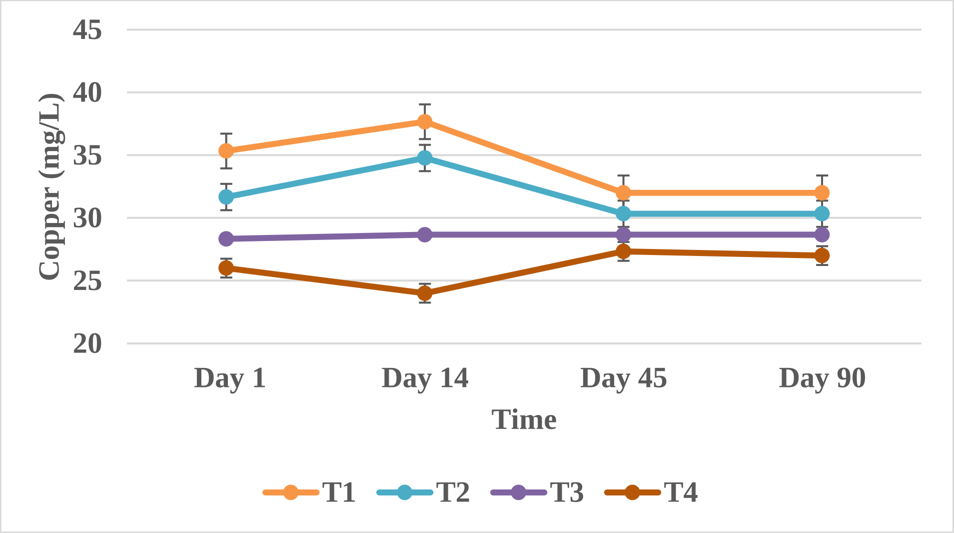

Figure 4.20 Mean variation of copper (Cu) concentration over time. Error bars represent standard error.

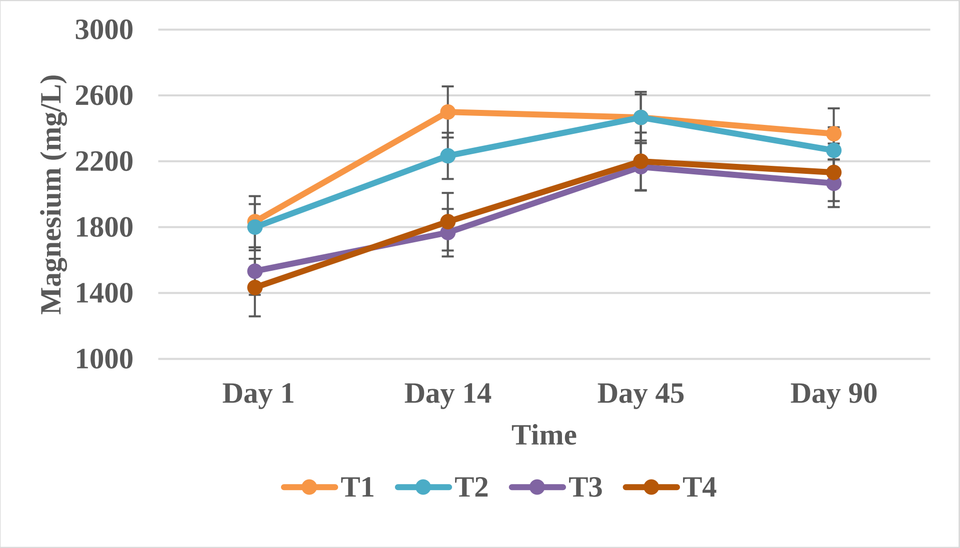

Figure 4.21 Mean variation of magnesium (Mg) concentration over time. Error bars represent standard error.

There was a significant difference in magnesium (Mg) over time (P = 0.01), with an increase of 28.13, 18.10, 53.66, and 47.166% from day 1 to 90 in T1 to T4, respectively. From day 1 to 45, in all treatments, Mg concentration increased, but after day 45, in all treatments, mean concentration reduced by an average of 1.68 ±1.25%.

Iron (Fe) amounts differed significantly over time in all treatments ($P < 0.001$); from day 1 to 90, Fe means increased by 38.06, 27.44, 66.34, and 56.54% in T1 to T4, respectively. As shown in Figure 4.22, the initial Fe mean values of T3 and T4 are significantly lower than T1 and T2. This may be due to the initial substrate material WS having a low Fe amount (109 mg/L) compared to PS (1350 mg/L), T3 and T4 have six times lower PS than T1 and T2. Studies done by Hazarika and Khwairahpam (2018) also reported the same pattern of observations. The initial Fe concentration gradually increases at the end of the experiment, which is explained by organic matter loss and mineralization. Elvira et al. (1998) also reported an increase in Fe amount in all treatments, but their reported values are higher (5400-9600 mg/L).

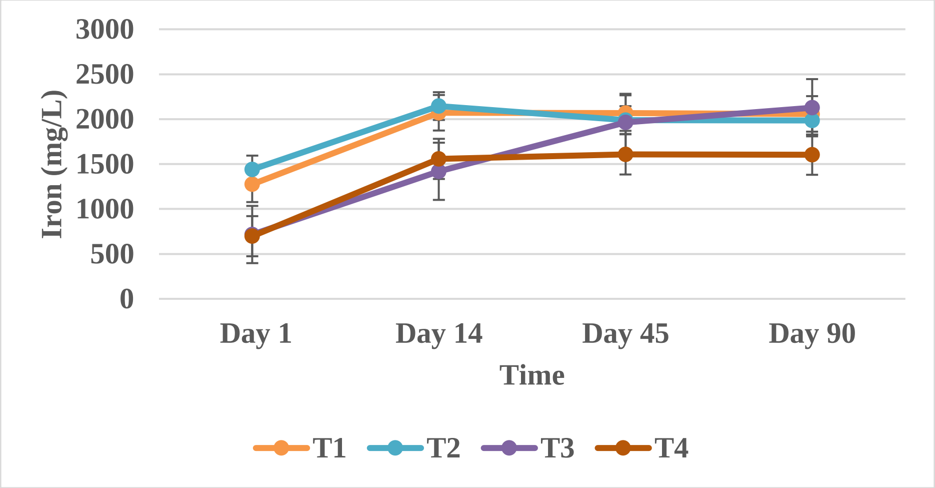

Figure 4.22 Mean variation of Iron (Fe) concentration over time, Error bars represent standard error.

A significant difference was observed in Manganese (Mn) over time ($P < 0.001$), with a mean increase of 370.83 mg/L in the first 14 days ($P = 0.001$) in all the treatments as shown in Figure 4.23. As shown in Figure 4.23, except for T2, the rest of the treatments showed a gradual increase in Mn concentration over time. In T2, from day 14 to day 45, there was a decrease of

11.78% in the mean concentration. From day 1 to day 90, a mean increase of 28.13, 18.10, 53.65, and 47.1% was observed in T1 to T4, respectively. Mn concentrations observed here showed higher values than observed by other studies with PS done by Elvira et al. (1998) and Hazarika and Khwairakpam (2018).

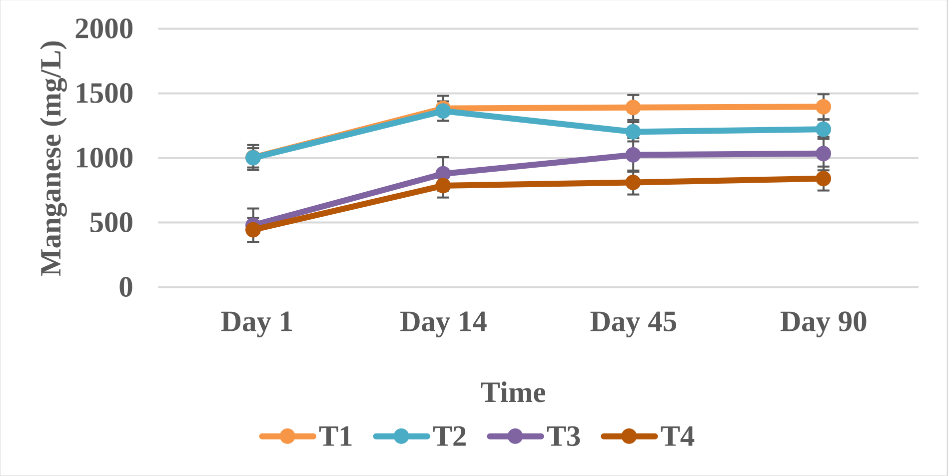

Figure 4.23 Mean variation of manganese (Mn) concentration over time. Error bars represent standard error.

There was a significant difference in sodium (Na) means over time ($P < 0.01$), and an increase in means of 820 mg/L ($P < 0.01$) was observed in the first 20 days. From day 1 to day 90, Na means concentration increased by 19.35, 36.44, 79.45, and 55.84% in T1 to T4, respectively (Figure 4.24). On day 1, there were no significant differences among treatments for Na concentration ($P = 0.45$), while on day 14, T1 significantly differed from the rest of the treatments ($P = 0.76$). On day 45 (T3, mean = 2687, $P = 0.02$) and day 90 (T3, mean = 2647, $P = 0.02$) T3 showed significantly higher mean values than T1, T2 and T4. An increase in Na concentration with time during composting was observed in the previous studies and explained to be due to the loss of overall dry mass during composting process (Hazarika and Khwairakpam, 2018)

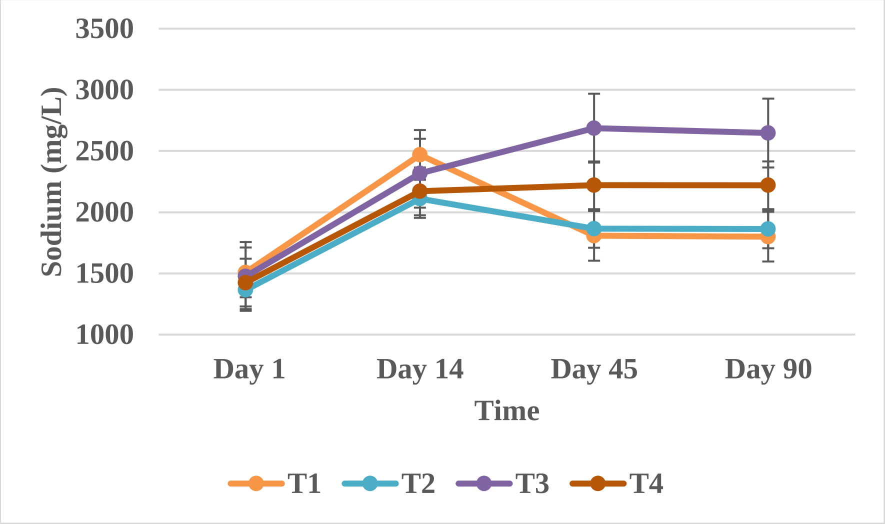

Figure 4.24 Mean variation of sodium (Na) concentration over time. Error bars represent standard error.

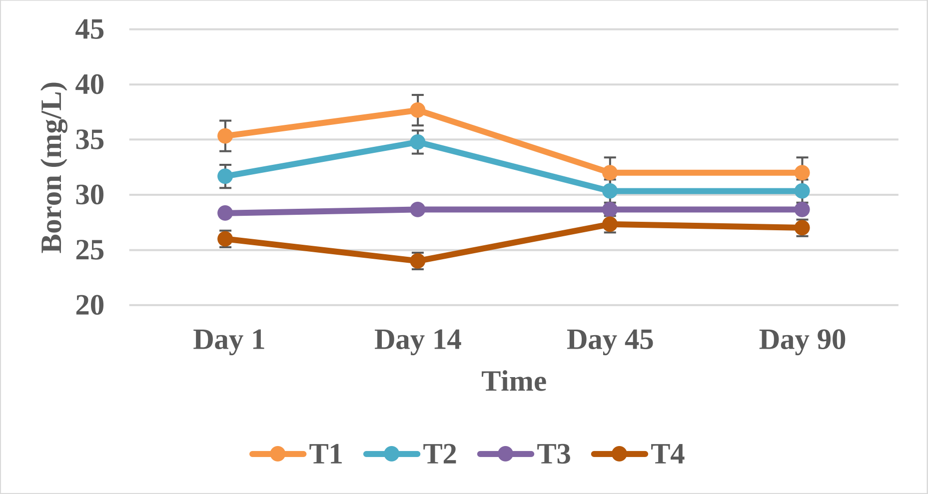

Figure 4.25 Mean variation of boron (B) concentration over time. Error bars represent standard error.

There was a significant difference in means observed over time in boron (B) ($P < 0.01$). From day 1 to day 90, concentration increased by 21.05, 33.78, 57.40, and 54.71 in T1 to T4, respectively. In all the treatments, there was no significant difference between day 45 and day 90 ($P = 0.75$). As shown in Figure 4.25, on both days 45 and 90, all of the treatments showed the same values for both days.

There was a significant difference over time in calcium (Ca) concentration (P = 0.02), except for T2, the other three treatments showed an increase in Ca mean concentration from day 1 to day 90 by 14.18, 24.70, and 35.79% in T1, T3, and T4 respectively (Figure 4.26). In all the treatments, there was no significant increase in Ca mean concentration after day 45 (P = 0.93). According to the studies done by Hazarika and Khwairakpam (2018), they also observed a gradual increase in Ca concentration due to dry mass loss.

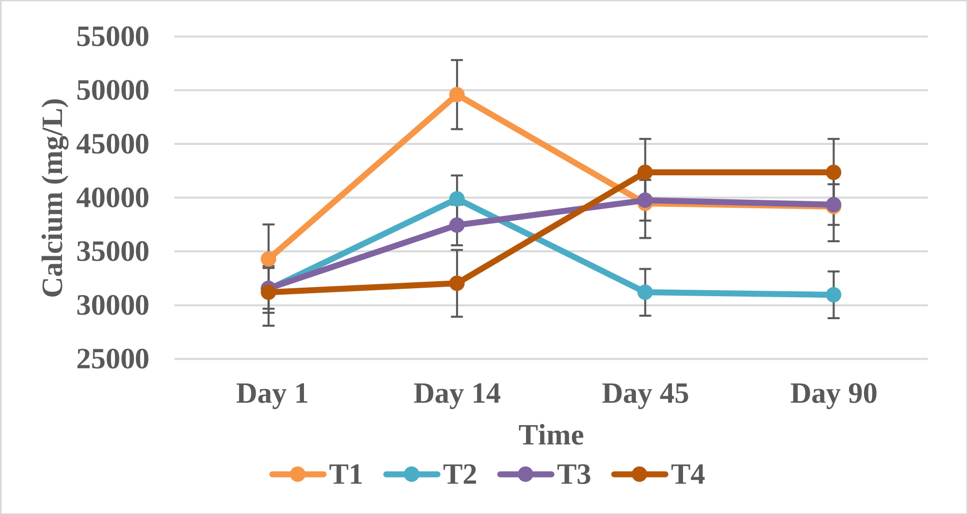

Figure 4.26 Mean variation of calcium (Ca) concentration over time. Error bars represent standard error.

According to Sullivan et al. (2018) and the Canadian Council of Ministers of the Environment (CCME), the guideline limit for compost (CCME, 2005), magnesium, and zinc levels are lower in value when compared to the recommended values over the whole period. However, all other nutrients fall within the recommended values (Table 2).

Table 4.1 Comparative physical and chemical properties of compost treatments before and after 90 days.

	T1		T2		T3		T4		
Properties	Day 1	Day 90	Day 1	Day 90	Day 1	Day 90	Day 1	Day 90	Recommendation
Ash (%)	73.2	54.7	61.4	45.2	60	36.8	47.7	43.9	
Organic matter (%)	81.8	80.1	85.2	82.2	84.9	83.5	87	84.3	<65
Moisture (%)	61.7	55.0	61.5	53.2	59.9	50.2	61.4	50.2	60-40
Total elements (mg/L)									
Phosphorous	5400	6367	4967	6000	4800	5800	4033	5067	3000-9000
Potassium	6067	9000	5667	9100	6033	8533	5567	8733	5000-15000
Calcium	34300	39167	31467	30967	31567	39367	31200	42367	15000-35000
Magnesium	**1833**	**2366**	**1800**	**2266**	**1533**	**2066**	**1433**	**2133**	**2500-7000**
Iron	1274	2058	1439	1983	745	2126	697	1604	n.a.[a]
Manganese	1003	1396	1001	1222	479	1034	444	841	n.a.
Copper	35	32	31	30	28	28	26	27	<400[b]
Zinc	**211**	**268**	**219**	**234**	**180**	**226**	**183**	**203**	**300-500**
Boron	25	30	24	33	18	28	17	27	n.a.
Sodium	1508	1800	1364	1862	1475	2648	1424	2220	<6000

T1 = paper sludge: wood shavings: poultry manure − 6:2:1, T2= paper sludge: wood shavings; poultry manure: biochar , T3=paper sludge: wood shavings: poultry manure −1:2:1, T4 = paper sludge: wood shavings: poultry manure: biochar. [a] Not available, [b] < represents the detection limit

According to one-way ANOVA, there is no significant difference (P = 0.325) among the treatments for OM variation over time. Higher OM, more than 65%, shows that compost may not have been composted thoroughly. It indicates that compost has more unstable OM that will be lost when applied to the field (Sullivan et al., 2018). All of the treatments show average values above 65%, providing evidence of incomplete composting.

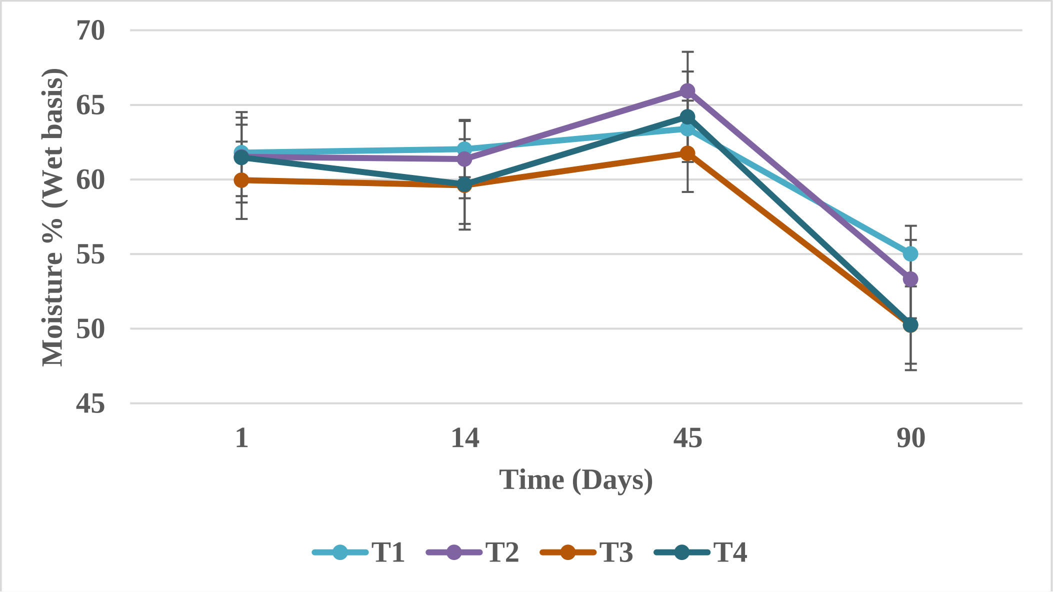

Figure 4.27 Mean values of moisture (%) change over time ratio changes over time. Error bars represent standard error.

Moisture is also a crucial factor for successful composting. During the initial stages, maintaining an adequate amount of moisture is vital for temperature distribution throughout the compost bin as well as for microbial activities. In this experiment, moisture was maintained at 60±2% during the initial stages. As shown in Figure 4.27, after 90 days, the moisture of the compost bins T1, T2, T3, and T4 dropped by 10.8, 13.3, 16.18, and 18.27%, respectively. According to one-way ANOVA, and Fisher pairwise comparison, there was no significant difference (P = 0.06) among the treatments for moisture variation from day 1 to 45. However, at the end of the composting process, there was a significant difference among treatments ($P = 0.02$) for moisture variation. T1 and T2 did not have any significant difference, while T1 is significantly different from T3 and T4 according to fisher pairwise comparison. Having more than 70% moisture level during the composting period creates anaerobic conditions inside compost piles, which trigger anaerobic digestion of organic materials by anaerobic bacteria. This is attributed to the production of CH_4 and N_2O during the process (Chowdhury et al., 2014; Chen et al., 2018). Final compost with 40-60% moisture is recommended (Sullivan et al., 2018). After the 90 days period, all treatments showed moisture content of 50-60%.

4.2.3 Greenhouse gas (GHG)

4.2.3.1 CO_2 and CH_4 emissions during composting

Microbial activity is one main reason for producing CO_2 and CH_4 during composting. The higher the CO_2 production, the higher the microbial activities, which indicates that rapid digestion of OM is happening in a compost pile (Chowdhury et al., 2014). On the other hand, higher CH_4 means there may be localized anaerobic conditions present in the compost pile (Chen et al., 2018). As shown in Figure 4.28 and Figure 4.29, CO_2 and CH_4 emissions rates are correlated with the temperature profile of all the treatments. A similar pattern was observed in the previous studies done by Chowdhury et al. (2014), and Chen et al. (2018), for CO_2 and CH_4 emissions. With the rise of the compost temperature after the second day of the experiment, CO_2 and CH_4 emissions from all the treatments also peaked during the thermophilic phase (from day 2 to 6) of the experiment, where the highest temperature was observed. After that, CO_2 and CH_4 emissions sharply decreased and maintained low values until day 90. For CO_2 in all treatments, day 3 showed the highest emission (Figure 4.28), while for CH_4, T1, and T2 showed the highest emission on day

4, and T3 and T4 showed the highest emission on day 3 (Figure 4.29). In both CO_2 and CH_4, T3 and T4 showed lower emissions than T1 and T2, which is explained by low temperature during the thermophilic phase (Figure 4.11).

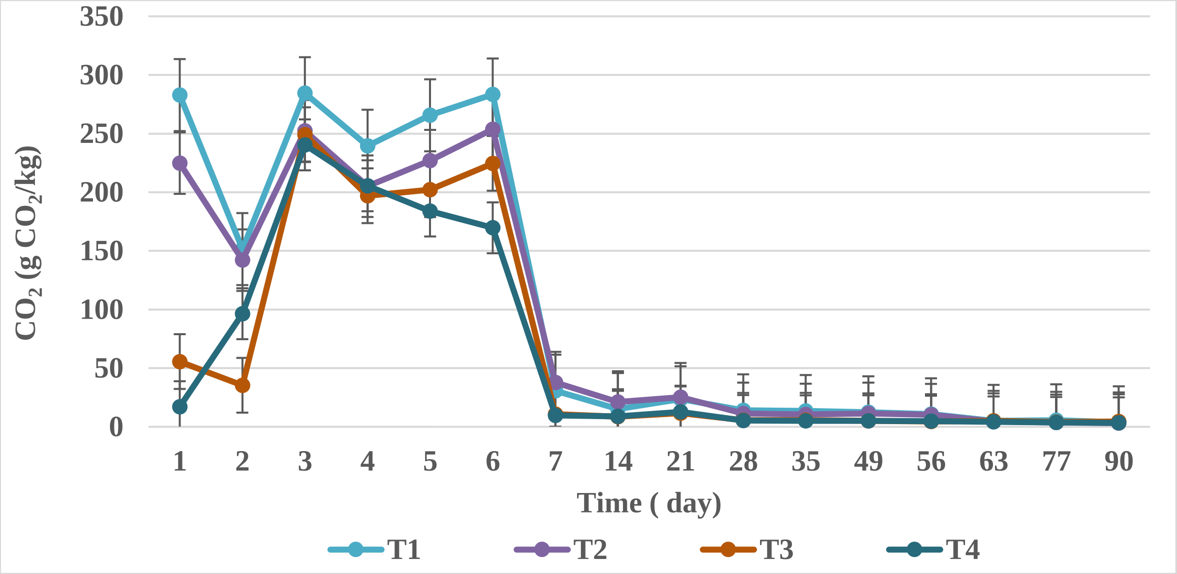

Figure 4.28 Carbon dioxide (CO_2) emission for all treatments during the composting. Error bars represent the standard error.

According to one-way ANOVA, during the first week of the experiment, there was a significant difference among the treatments for CO_2 ($P = 0.002$) and CH_4 ($P < 0.001$) emission. According to the Fishers pairwise comparison, T3 and T4 belong to the same mean group, while T1 and T2 belong to the same mean group. But there was no significant difference between treatments 2 and 3 for CO_2 emission. Additionally, for CH_4 emissions, all of the treatments significantly differed from each other.

CO_2 emission was an indicator of rapid OM degradation and microbial enzymatic activities (Zhang et al., 2014; Chen et al., 2018). As shown in Figure 4.28, the trend of CO_2 emission increased rapidly in the first few days and then decreased rapidly and continuously kept decreasing.

The peak values for CO_2 generation were 28.4, 25.33, 24.9, and 24.03 g/kg on day three from T1 to T4, respectively. T1 and T2 had higher rates of PS than T3 and T4 (3:1 by wet weight basis). The ratio between PS: WS in T1 and T2 was 3:1, while in T3 and T4, it was 1:2. According to the increase in temperature, CO_2 and CH_4 emissions in T1 and T2 showed higher values, this shows that there is a relationship between the mixing ratios of material for temperature increase and leading to higher CO_2 and CH_4 emissions. High-temperature increase is attributed to high microbial activity, which is associated to highly readily available C for microbial use.

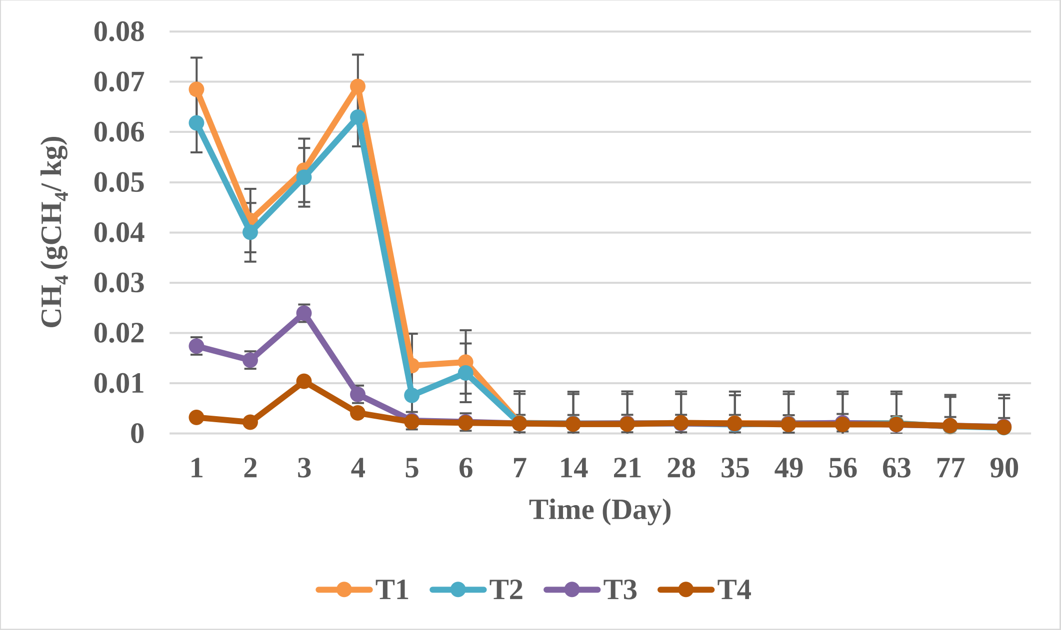

Figure 4.29 Methane (CH_4) emission for all treatments. Error bars represent the standard error.

This is also supported by a higher mass fraction of total organic matter (TOM) in the initial composting and the increase in temperature and increase of enzymatic activities (Chen et al., 2018). As shown in Figure 4.20, the TOM content of all the treatments decreases in the initial stages, and after 45 days, seem to have stable values with the exception of T1. This shows that the degradation of TOM was high during the initial stages of composting, which explains the emission of CO_2 and CH_4. T2 and T4 have BC (4%), but it did not show any significant effect on CO_2 emission during the experimentation process. As mentioned by Awasthi et al. (2020), higher amount of BC addition (10%) will increase CO_2 generation because of the acceleration of aeration shows an effect on organic waste degradation (Awasthi et al., 2020). A higher application of BC will increase the rate of degradation in compost due to the generation of porosity, which increases surface area and abiotic oxidation of BC (Awasthi et al., 2020; Godlewska et al., 2017).

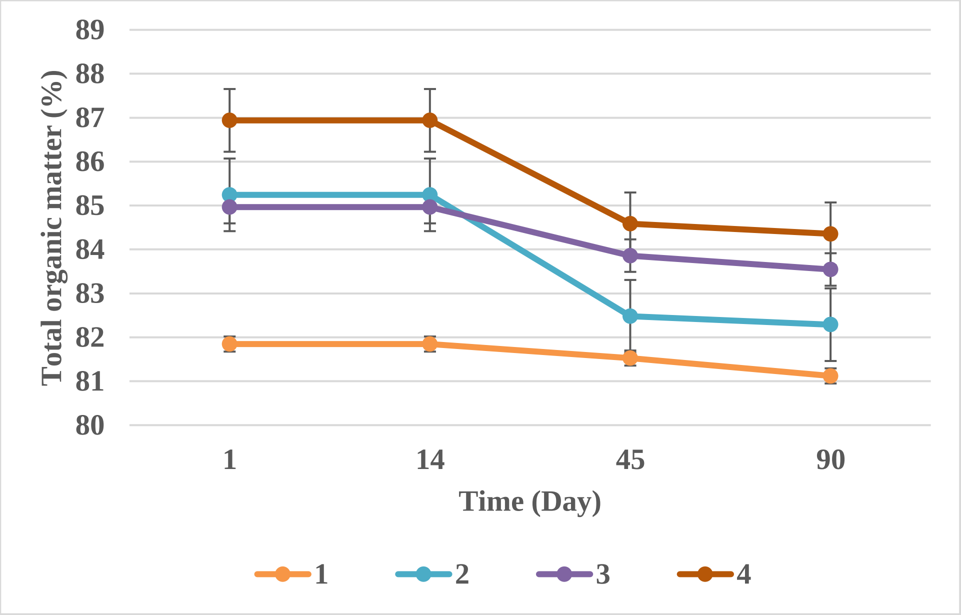

Figure 4.30 Total organic matter (TOM) variability over time for all treatments. Error bars represent the standard error.

In the initial stage of composting, high oxygen consumption by microbes, high CO_2 production, and temperature increase along with moisture release may limit oxygen diffusion thereby limiting oxygen availability in some areas in the compost pile. These anaerobic conditions trigger CH_4 production due to the anaerobic digestion of OM by anaerobic microorganisms. When compared with the other experiments related to composting done by Chen et al. (2018), Chowdhury et al. (2014), and Awasthi et al. (2020) rate of CH_4 emissions is comparatively low in this experiment. This can be attributed to continuous aeration during the experimentation period and maintaining moisture levels at 60% during the initial stages, which prevented the occurrence of severe anaerobic conditions inside the bins.

Also, treatments with BC (T2 and T4) have lower mean values for CH_4 emission in comparison to T1 and T3. According to Lin et al. (2022), BC inhibits CH_4 emission by increasing the activities of Acetyl-CoA synthase and β-glucosidase and decreasing the activities of coenzymes related to CH_4 production. After day 5, for both CO_2 ($P = 0.706$) and CH_4 ($P = 0.106$) emissions, there was no significant difference among treatments according to one-way ANOVA.

4.2.3.2 N_2O emissions

During composting, N_2O can be produced in both anaerobic and aerobic conditions (Chowdhury et al., 2014). According to the literature (Awasthi et al., 2018; Chen et al., 2018), N_2O plays an important role in N losses during composting and contributes to global warming.

N_2O is mainly produced by the nitrification of ammonium nitrogen and denitrification of nitrate nitrogen (Chen et al., 2018). As shown in Figure 4.31, at the initial stages, N_2O emission shows peak values from day 2 to 7 for all the treatments; during the thermophilic stage of composting, both aerobic and anaerobic bacteria coexisting contribute to the emission of N_2O (Chowdhury et al., 2014). Based on the observation this high emission rate may be due to the incomplete denitrification or nitrification process that occurred in the compost bins. According to the two-sample t-test during the first five days, there is a significant difference among the treatments for N_2O emission; T3 is significantly different from T1 and T2. T4 is significantly different from T1 and T2. While there is no significant difference between T1 to T2 and T3 to T4.

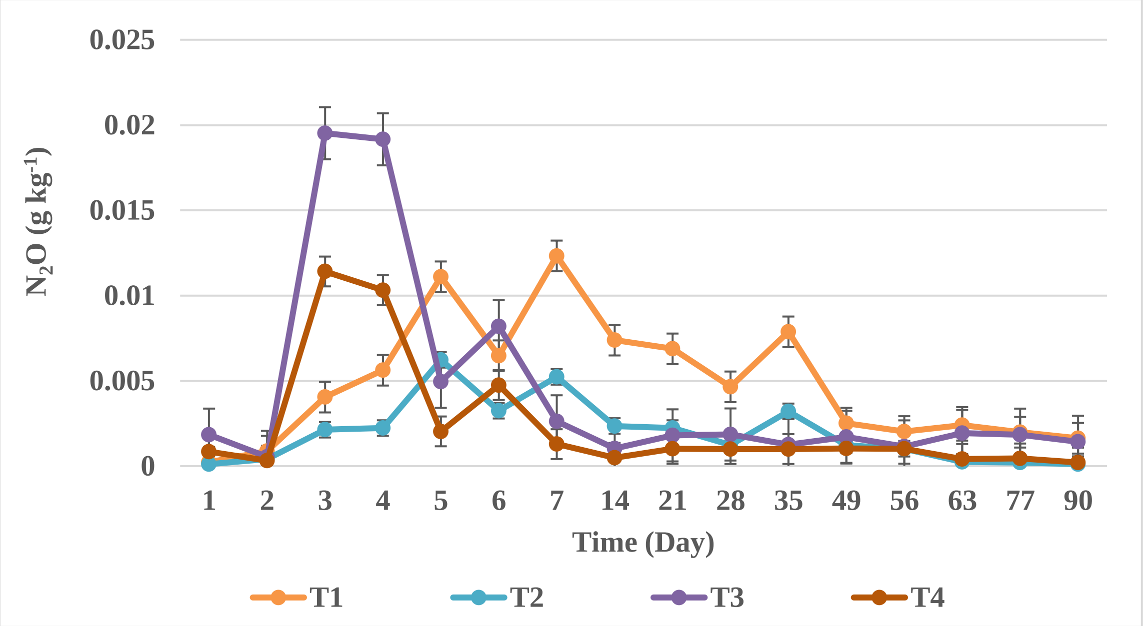

Figure 4.31 Nitrous dioxide (N_2O) emission for all treatments. Error bars represent the standard error.

Table 4.2. Results from two-way t-test for mean comparison of N_2O emission during first five days.

Comparison	P-value	Mean
T1-T2	0.871	2.69^b, 2.53^b
T1-T3	0.021	2.69^b, 10.27^a
T1-T4	0.035	2.69^b, 8.59^a
T2-T3	0.018	2.53^b, 10.27^a
T2-T4	0.031	2.53^b, 8.59^a
T3-T4	0.307	10.27^a, 8.59^a

Mean values that share the same letter are not significantly different at 0.05 significant level.

After day 7, from day 14 to 49, there was a significant difference between T1 and T2 to T3 and T4 ($P < 0.001$) according to one-way ANOVA Fisher's pairwise comparison. T1 and T2 showed the highest mean values (5.911, 4.11), while T3 and T4 had lower mean values (1.486, 1.456). But there is no significant difference among BC-added treatments to its comparative treatment though they showed low values. T2 and T4 were BC added to T1 and T3, respectively. As shown in Figure 4.21, T2 has lower mean values for N_2O emission than T1, and T4 has lower emission rates than T3. This shows that there is some effect of BC on N_2O emission. The highest peak for N_2O was observed on day 3 for T3 and T4, while T3 showed the highest peak, an average of 0.0019% on day 3 for all treatments. T1 and T2 showed the highest peak on days 7 and 5, respectively. BC inhibited N_2O emission and accelerated the release period (Awasthi et al., 2020). During the initial stages of composting, high NH_4^+-N cause for emission and production of N_2O due to both biological nitrification and denitrification (Chowdhury et al., 2014). As shown in Figure 4.14a, T3 has a high NH_4^+-N concentration during the initial stage, which may be the reason for high N_2O emissions during the initial days of composting in T3. But when compared with the other similar composting studies carried out by Chen et al. (2018), Chowdhury et al. (2014), and Awasthi et al. (2020) also have seen a similar pattern for N_2O emission. But the rates are much higher than those found in this experiment.

Even though the mechanism of N_2O emission during composting is still controversial in the existing results, according to the literature (Chan et al., 2016; Chen et al., 2018) the form of N in compost material, size of reactors, aeration, and moisture content may have a significant effect on N_2O emission. Literature shows that the effect of BC on mitigating GHG emissions is significantly related to the rate of addition of BC (Awashi et al., 2017; Yin et al., 2021). Most studies which added different BC rates changing from 0–20% (w/w) showed that 10% was the most efficient (Awasthi et al., 2020). According to Awasthi et al. (2020), the addition of 8–18% BC reduces N_2O, NH_4, and CH_4 emissions by 95.1–97.2%, 58.0–65.1%, and 92.8–94.3%, respectively. A low BC amount, around 2–6%, had little effect on GHG and NH_3 emissions. In both experiments, 4% of BC was added on a dry mass basis, showing little to no effect on GHG and NH_3 emissions. At the farmer level of large-scale compost production, an addition of 10% BC may become a limiting factor due to an association with high market price of BC in NL. Spending a large sum of money to buy raw materials is not a sustainable and effective solution for farmers.

4.2.4 Compost maturity test for compost produced from the bin experiment

Compost stability and maturity should be tested before the application of compost to agricultural lands. Unstable immature compost adversely affects seed germination, plant health, and the soil environment due to the phytotoxic substance and available N (Luo et al., 2018). To assay the toxic substances in immature compost germination index (GI) was used (Luo et al., 2018). GI has a positive correlation with humification (Gavilance-Teran et al., 2016), while Guo et al. (2021) have shown that it is negatively correlated with the NH_4^+-N content of compost. The GI is one of the most broadly accepted tests for compost quality. According to Tiquia et al. (1996), compost that has a GI of more than 80% is suggested to be phytotoxic free compost, which is suitable for land application. The GI of all the treatments at day 90 shows GI values that are greater than 80% (Figure 4.32). Therefore, the compost produced shows no phytotoxicity.

Another factor that is used to detect compost maturity is the high temperature during the thermophilic phase, along with its duration and stability. This is important for the sanitization of compost. During the experiment, none of the compost bins reached temperatures above 50°C. This shows they did not reach the required temperature during the thermophilic phase in order to destroy pathogens and weed seeds. An average of 48°C in all treatments were observed during the thermophilic phase of the experiment, lasting for three days from day 2 to day 4, which is very short. The main reason for low-temperature observation is possibly connected to the lack of substantial enough materials to facilitate microbial activities (Goyal et al., 2005). The compost bin with a 24.5 L capacity did not have enough materials to sustain the temperature for a longer period, as suggested by the University of Arkansas Division of Agriculture Research and Extension Service (https://www.uaex.uada.edu/publications/PDF/FSA-6036.pdf). Compost bins should be at least one cubic yard to ensure good thermophilic compost. Also, PS and WS have low readily available C, which makes it difficult for microorganisms to start their digestion and continue it. Also, a lack of enough inoculum could be another reason for less microbial activity, resulting in a lower overall temperature increase.

Another indicator that shows mature compost is the final C/N of compost. According to the literature, compost with a 12–15:1 C/N is good for plant application (Sun et al., 2014; Sullivan et al., 2018). During the experiment, compost only reached a C/N of an average of 24, 24, 28, and 26 for T1 to T4, respectively. This shows that complete composting of organic matter may not have happened during the 90-day period under the testing conditions of this experiment. This was also observed by Charest and Beauchamp (2002) in their experiment done to de-ink paper sludge with PM composting. As the authors suggested, decomposing PS needs more than 24 weeks of composting. Also, some suggested using other N-rich substrates, such as dairy sludge or green manure (Elvira et al., 1998; Hazarika and Khwairakpam, 2018).

4.2.5 Germination index (GI)

GI is used to test the phytotoxicity of compost. There was no significant difference among treatments during days 1, 14, and 45, according to one-way ANOVA (P = 0.676). But samples collected on day 90 show a significant difference between T1 and T2 to T3 and T4 (P = 0.004) according to fisher pairwise comparisons. T3 and T4 show higher GI than T1 and T2 (Figure 4.32). According to the literature, if the GI ≥ 80%, it is considered a phytotoxicity-free compost (Tiquia et al., 1996). It shows all treatments have reached that level. A slight GI drop on day 14 may be due to ammonium collection by organic matter degradation.

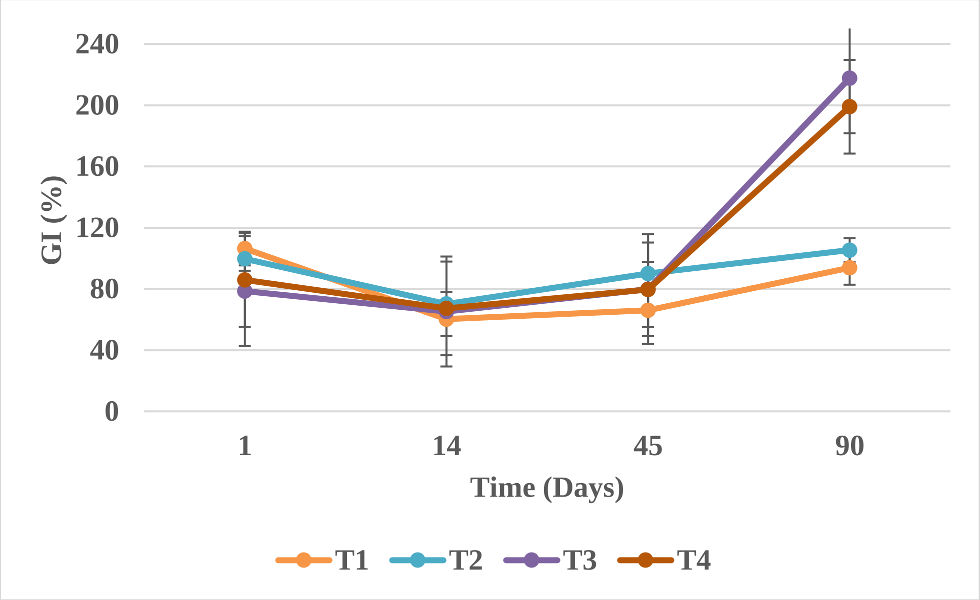

Figure 4.32 Average germination index (GI) changes with time during the compost experiment. Error bars represent standard error.

CHAPTER 5: Conclusions

Treatments with higher paper sludge (PS) ratio showed higher CO_2 and CH_4 emissions, lower N_2O emissions, and the highest average temperature during the thermophilic phase compared to treatments with lower PS ratio. Additionally, the final compost from the higher PS ratio treatments has lower C/N. These findings lead to the conclusion that treatment with higher PS ratio were the most suitable mixing ratio for composting. These laboratory-scale experiments showed that the compost thermophilic phase increase in temperature has an effect on carbon dioxide (CO_2), methane (CH_4), and nitrous dioxide (N_2O) emissions.

Poultry manure (PM) is a good source of nitrogen (N), which can be used to balance the C/N of PS as a readily available N source for microbial digestion of compost. Despite not reaching sufficiently matured C/N for field use, the composting experiments conducted during this research show the potential of using PS in combination with other natural resource by-products available in Newfoundland and Labrador (NL) to produce nutrient-rich organic fertilizer. This compost can serve as an alternative to inorganic fertilizer, but may not fully replace the use of inorganic fertilizer in intensive farming. Using fishpond waste as raw material for composting with PS did not prove feasible due to the high initial moisture content (92%) and high volatilization of N.

The incubation study showed that the addition of wood ash (WA) did not have an effect on pH when composting PS with PM. Overall, greenhouse gas (GHG) emissions observed in this experiment do not show any significant differences among the treatments.

The addition of 4% biochar (BC) did not show any noticeable effect on reducing any GHG or ammonia (NH_3) emissions. It showed some effect on controlling CH_4 and N_2O emissions, but that effect was not statistically significant. It was also observed that CH_4 emissions during this experiment were much lower than the similar experiments done previously. This indicates that continuously forced aeration and maintaining moisture around 60% have an effect on reducing CH_4 emissions.

Future studies need to be done with large-scale outdoor composting experiments to check GHG fluctuation in large-scale compost piles with these mixing ratios. Applying biochar at a higher range (5–10%) may give better results but may not be sustainable for practical application.

Also, because under the tested conditions 90 days were not enough for composting PS and WS, the time period of the experiment needs to be lengthened. Further, introducing inoculants from outside sources (like commercially available inoculation products) can be another option for quick decomposition, but the potential effects of such modifications on GHG emissions will need to be assessed.

According to the provincial action plan titled "The way forward on climate change in NL", initiatives have been launched to lower GHG emissions and promote the use of organic fertilizers in agriculture, aiming for sustainable economic growth within the province. These projects, specifically focus on reducing GHG emissions from large organic waste collection sites such as Corner Brook Pulp and Paper Limited (CBPPL). The findings of this research study can serve as a valuable guide in addressing potential challenges related to waste management at the CBPPL. Implementing the findings can provide significant benefits to the local communities in NL.

When manufacturing compost within the province, it is necessary to follow the provincial guidelines issued by the Department of Environment and Climate Change of the government of NL. In addition, compliance with the compost quality guidelines provided by the Canadian Council of Ministers of Environment (CCME) for its use as fertilizer and T-4-120 regulation of the "compost under the fertilizer" act administered by the Canadian Food Inspection Agency (CFIA) for packing and selling compost products is also necessary. Before adopting the compost production methodology proposed in this research study, further steps need to be taken. These include conducting additional laboratory analysis, field trials, and plant trials under controlled environments (greenhouse) to assess the effect of this compost on plant growth (plant bioassays). Additionally, both composts and plants should undergo laboratory analysis to check any potential contaminants like heavy metals from PS and PM, and antibiotic and pathogenic contaminants from

PM, which may pose risks to human health. Taking these precautions will ensure that the compost production process aligns with the required standards and safeguards the well-being of the environment and the community.

Furthermore, special attention must be given to community involvement. Whether for farmlands or households, a solid understanding of the composting process is needed to effectively utilize locally available raw materials for compost production. To achieve this, conducting a social assessment and community outreach becomes essential in disseminating information and gathering feedback. Alternatively, communities can consider devising plans to establish large-scale compost facilities as a business venture, enabling them to commercialize their product for sale. However, before implementing such large-scale compost production, it is necessary to conduct a field experiment using large compost piles with predetermined mixing ratios. This experiment will allow for the monitoring of GHG and NH_3 emissions, comparing the observed effects with other PS management practices like incineration and landfilling. The data collected will aid in calculating the carbon footprint and GHG emissions providing a clearer understanding of the most effective approach to managing this industrial by-product.

CHAPTER 6: References

A&L Laboratories. (2004). Compost Management. https://alcanada.com/pdf/technical/compost/Compost_Handbook.pdf

Agricultural Greenhouse Gas Indicator - agriculture.canada.ca. (2021, June 25). https://agriculture.canada.ca/en/agricultural-production/agricultural-greenhouse-gas-indicator

Amous, S. (1999). The role of wood energy in Africa. In: Rivero SI (ed) Wood energy today for tomorrow. Food and Agriculture Organization of the United Nations, Rome. https://www.fao.org/3/x2740E/x2740e00.htm#acro

Aljameel, K. M., Omage, J. J., Olugbemi, T. S., & Usman, A. A. (2021). Chemical and microbial analysis of bedding material (wood shavings) treated with graded levels of aluminium sulphate (Alum). *Archives of Current Research International*, 21(6), 32-41.

Akdeniz, N. (2019). A systematic review of biochar use in animal waste composting. *Waste Management*, 88, 291-300.

Ali, M. H., Sobze, J.-M., Pham, T. H., Nadeem, M., Liu, C., Galagedara, L., Cheema, M., & Thomas, R. (2020). Carbon nanoparticles functionalized with carboxylic acid improved the germination and seedling vigor in upland boreal forest species. *Nanomaterials* (Basel, Switzerland), 10(1), 176. https://doi.org/10.3390/nano10010176

Appiah-Effah, E., Nyarko, K. B., Antwi, E. O., & Awuah, E. (2016). Effect of bulking materials and mixing ratios on concentration of nutrients during composting of raw fecal sludge from peri-urban areas. *Water Practice and Technology*, 11(1), 234–242. https://doi.org/10.2166/wpt.2016.028

Awasthi, M. K., Wang, Q., Chen, H., Wang, M., Ren, X., Zhao, J., Li, J., Guo, D., Li, D., Awasthi, S. K., Sun, X., & Zhang, Z. (2017). Evaluation of biochar amended biosolids co-composting to improve the nutrient transformation and its correlation as a function for the production of nutrient-rich compost. *Bioresource Technology*, 237, 156–166. https://doi.org/10.1016/j.biortech.2017.01.044

Awasthi, M. K., Pandey, A. K., Bundela, P. S., & Khan, J. (2015). Co-composting of organic fraction of municipal solid waste mixed with different bulking waste: Characterization of physicochemical parameters and microbial enzymatic dynamic. *Bioresource Technology*, 182, 200–207. https://doi.org/10.1016/j.biortech.2015.01.104

Awasthi, M. K., Duan, Y., Awasthi, S. K., Liu, T., & Zhang, Z. (2020). Influence of bamboo biochar on mitigating greenhouse gas emissions and nitrogen loss during poultry manure composting. *Bioresource Technology*, 303, 122952–122952. https://doi.org/10.1016/j.biortech.2020.122952

Ayilara, M. S., Olanrewaju, O. S., Babalola, O. O., & Odeyemi, O. (2020). Waste Management through Composting: Challenges and Potentials. *Sustainability* (Basel, Switzerland), 12(11), 4456. https://doi.org/10.3390/su12114456

Bang-Andreasen, T., Nielsen, J. T., Voriskova, J., Heise, J., Rønn, R., Kjøller, R., Hansen, H. C. B., & Jacobsen, C. S. (2017). Wood ash induced pH changes strongly affect soil bacterial numbers and community composition. *Frontiers in Microbiology*, 8, 1400–1400. https://doi.org/10.3389/fmicb.2017.01400

Barral, M.T., & Paradelo, R. (2011). A review on the use of phytotoxicity as a compost quality indicator. *Dynamic Soil, Dynamic Plant* 5(2), 36-44.

Batham, M., Gupta, R., & Tiwari, A. (2013). Bioremediation & biodegradation. Beauchamp, C. J., Charest, M.-H., & Gosselin, A. (2002). Examination of environmental quality of raw and composting de-inking paper sludge. *Chemosphere* (Oxford), 46(6), 887–895. https://doi.org/10.1016/S0045-6535(01)00134-5

Beauchamp, C. J., Charest, M.-H., & Gosselin, A. (2002). Examination of environmental quality of raw and composting de-inking paper sludge. *Chemosphere* (Oxford), 46(6), 887–895. https://doi.org/10.1016/S0045-6535(01)00134-5

Bernal, M. P., Alburquerque, J. A., & Moral, R. (2009). Composting of animal manures and chemical criteria for compost maturity assessment. A review. *Bioresource Technology*, 100(22), 5444–5453. https://doi.org/10.1016/j.biortech.2008.11.027

Bernal, M. P., Sommer, S. G., Chadwick, D., Qing, C., Guoxue, L., & Michel, F. C. (2017). Current approaches and future trends in compost quality criteria for agronomic, environmental, and human health benefits. *Advances in Agronomy*, 144, 143–233. https://doi.org/10.1016/bs.agron.2017.03.002

Bougnom, B. P., Mair, J., Etoa, F. X., & Insam, H. (2009). Composts with wood ash addition: A risk or a chance for ameliorating acid tropical soils. *Geoderma*, 153(3), 402–407. https://doi.org/10.1016/j.geoderma.2009.09.003

Brown, S., Kruger, C., & Subler, S. (2008). Greenhouse gas balance for composting operations. *Environmental Quality*, 37(4), 1396-1410.

Byakodi, A. S., & Babu, B. T. S. (2022). Aerobic Composting of sugar pressmud with stabilized spentwash and selected microbial consortium. *Eurobiotech*, 6(1), 32–39. https://doi.org/10.2478/ebtj-2022-0004

Cabral, F., Vasconcelos, E., Goss, M., & Cordovil, C. M. d. (1998). Value, use, and environmental impacts of pulp-mill sludge additions to forest and agricultural lands in Europe. *Environmental Reviews*, 6(1), 55–64. https://doi.org/10.1139/a98-004

CCME (2005). Guidelines for Compost Quality.

https://ccme.ca/en/res/compostgdlns_1340_e.pdf

Cha, J. S., Park, S. H., Jung, S. C., Ryu, C., Jeon, J. K., Shin, M. C., & Park, Y. K. (2016). Production and utilization of biochar: A review. *Industrial and Engineering Chemistry*, 40, 1-15.

Chan, M. T., Selvam, A., & Wong, J. W. C. (2016). Reducing nitrogen loss and salinity during "struvite" food waste composting by zeolite amendment. *Bioresource Technology*, 200, 838–844. https://doi.org/10.1016/j.biortech.2015.10.093

Chang, J. I., & Chen, Y. J. (2010). Effects of bulking agents on food waste composting. *Bioresource Technology, 101*(15), 5917–5924. https://doi.org/10.1016/j.biortech.2010.02.042

Chang, Y., & Hudson, H. J. (1967). The fungi of wheat straw compost. *Ecological Studies,* 50(4), 649-666.

Charest, M. H., & Beauchamp, C. (2002). Composting of de-inking paper sludge with poultry manure at three nitrogen levels using mechanical turning: behavior of physico-chemical parameters. *Bioresource Technology*, 81(1), 7–17. https://doi.org/10.1016/S0960-8524(01)00104-3

Chen, Y. (2003). Nuclear magnetic resonance, infra-red and pyrolysis: application of spectroscopic methodologies to maturity determination of composts. *Compost Science & Utilization,* 11(2), 152-168.

Chen, Y., Liu, Y., Li, Y., Wu, Y., Chen, Y., Zeng, G., Zhang, J., & Li, H. (2017). Influence of biochar on heavy metals and microbial community during composting of river sediment with agricultural wastes. *Bioresource Technology*, 243, 347–355. https://doi.org/10.1016/j.biortech.2017.06.100

Chen, H., Awasthi, M. K., Liu, T., Zhao, J., Ren, X., Wang, M., Duan, Y., Awasthi, S. K., & Zhang, Z. (2018). Influence of clay as additive on greenhouse gases emission and maturity evaluation during chicken manure composting. *Bioresource Technology*, 266, 82–88. https://doi.org/10.1016/j.biortech.2018.06.073

Chen, H., Awasthi, S. K., Liu, T., Duan, Y., Ren, X., Zhang, Z., & Awasthi, M. K. (2020). Effects of microbial culture and chicken manure biochar on compost maturity and greenhouse gas emissions during chicken manure composting. *Hazardous Materials*, 389, 121908.

Chowdhury, M. A., de Neergaard, A., & Jensen, L. S. (2013). Potential of aeration flow rate and bio-char addition to reduce greenhouse gas and ammonia emissions during manure composting. *Chemosphere* (Oxford), 97, 16–25. https://doi.org/10.1016/j.chemosphere.2013.10.030

Chowdhury, M. A., de Neergaard, A., & Jensen, L. S. (2014). Composting of solids separated from anaerobically digested animal manure: Effect of different bulking agents and mixing ratios on emissions of greenhouse gases and ammonia. *Biosystems Engineering*, 124, 63–77. https://doi.org/10.1016/j.biosystemseng.2014.06.003

Chuan-kuan, W., Feldkirchner, D. C., Gower, S. T., Ferris, J., & Kruger, E. L. (2005). Effects of amendments of paper mill sludge and nutrients on soil surface CO_2 flux in northern hardwood forests. *Journal of Forestry Research*, 16(4), 265–269. https://doi.org/10.1007/BF02858186

Dinel, H., Marche, T., Schnitzer, M., Paré, T., & Champagne, P. (2004). Co-composting of paper mill sludge and hardwood sawdust under two types of In-vessel processes. *Environmental Science and Health. Part B, Pesticides, Food Contaminants, and Agricultural Wastes*, 39(1), 139–151. https://doi.org/10.1081/PFC-120027445

Dróżdż, D., Malińska, K., Kacprzak, M., Mrowiec, M., Szcypior, A., Postawa, P., & Stachowiak, T. (2020). Potential of fishpond sediments composts as organic fertilizers. *Waste Biomass*,11, 5151–5163. https://doi.org/10.1007/s12649-020-01074-6

Dhamodharan, K., Varma, V. S., Veluchamy, C., Pugazhendhi, A., & Rajendran, K. (2019). Emission of volatile organic compounds from composting: A review on assessment, treatment and perspectives. *The Science of the Total Environment*, 695, 133725–133725. https://doi.org/10.1016/j.scitotenv.2019.133725

Ekinci, K., Keener, H. M., & Elwell, D. L. (2000). Composting short paper fiber with broiler litter and additives. *Compost Science & Utilization*, 8(2).

El-Naggar, A., Lee, S. S., Rinklebe, J., Farooq, M., Song, H., Sarmah, A. K., Zimmerman, A. R., Ahmad, M., Shaheen, S. M., & Ok, Y. S. (2019). Biochar application to low fertility soils: A review of current status, and future prospects. *Geoderma*, 337, 536–554. https://doi.org/10.1016/j.geoderma.2018.09.034

Elvira, C., Sampedro, L., Benítez, E., & Nogales, R. (1998). Vermicomposting of sludges from paper mill and dairy industries with Eisena andrei: A pilot-scale study. *Bioresource Technology*, 63(3), 205–211. https://doi.org/10.1016/S0960-8524(97)00145-4

Ermolaev, E., Jarvis, Å., Sundberg, C., Smårs, S., Pell, M., & Jönsson, H. (2015). Nitrous oxide and methane emissions from food waste composting at different temperatures. *Waste Management*, 46, 113-119.

Fabrice, L. Y., Veronica, E. M., Ngwa, M. N., David, T. T., & Esongami, N. E. (2021). Variability of physiochemical properties of livestock manure with added wood shavings during Windrow Composting. *Environmental Science & Technology*, 15(2), 117–123. https://doi.org/10.5897/AJEST2020.2973

Friedrich, E., & Trois, C. (2011). Quantification of greenhouse gas emissions from waste management processes for municipalities–A comparative review focusing on Africa. *Waste Management*, 31(7), 1585-1596.

Faubert, P., Barnabé, S., Bouchard, S., Côté, R., & Villeneuve, C. (2016). Pulp and paper mill sludge management practices: What are the challenges to assess the impacts on greenhouse gas emissions. *Resources, Conservation and Recycling*, 108, 107–133. https://doi.org/10.1016/j.resconrec.2016.01.007

Fidel, R. B., Laird, D. A., Thompson, M. L., & Lawrinenko, M. (2017). Characterization and quantification of biochar alkalinity. *Chemosphere*, 167, 367–373.

Gagnon, B., Simard, R. R., Lalande, R., & Lafond, J. (2003). Improvement of soil properties and fruit yield of native lowbush blueberry by papermill sludge addition. *Canadian Journal of Soil Science*, 83(1), 1-9.

Gagnon, B., & Ziadi, N. (2012). Papermill biosolids and alkaline residuals affect crop yield and soil properties over nine years of continuous application. *Canadian Journal of Soil Science*, 92(6), 917–930. https://doi.org/10.1139/CJSS2012-02

Gavilanes-Terán, I., Jara-Samaniego, J., Idrovo-Novillo, J., Bustamante, M. A., Moral, R., & Paredes, C. (2016). Windrow composting as horticultural waste management strategy – A case study in Ecuador. *Waste Management* (Elmsford), 48, 127–134. https://doi.org/10.1016/j.wasman.2015.11.026

Godlewska, P., Schmidt, H. P., Ok, Y. S., & Oleszczuk, P. (2017). Biochar for composting improvement and contaminants reduction. A review. *Bioresource Technology*, 246, 193-202.

Goyal, S., Dhull, S. K., & Kapoor, K. K. (2005). Chemical and biological changes during composting of different organic wastes and assessment of compost maturity. *Bioresource Technology*, 96(14), 1584–1591. https://doi.org/10.1016/j.biortech.2004.12.012

Gupta, R., & Garg, V. K. (2008). Stabilization of primary sewage sludge during vermicomposting. *Hazardous Materials*, 153(3), 1023–1030. https://doi.org/10.1016/j.jhazmat.2007.09.055

Guo, R., Li, G., Jiang, T., Schuchardt, F., Chen, T., Zhao, Y., & Shen, Y. (2012). Effect of aeration rate, C/N ratio and moisture content on the stability and maturity of compost. *Bioresource Technology*, 112, 171–178. https://doi.org/10.1016/j.biortech.2012.02.099

Hackett, G. A., Easton, C. A., & Duff, S. J. (1999). Composting of pulp and paper mill fly ash with wastewater treatment sludge. *Bioresource Technology*, 70(3), 217–224. https://doi.org/10.1016/S0960-8524(99)00048-6

Hao, X., Chang, C., & Larney, F. (2004). Carbon, nitrogen balances and greenhouse gas emission during cattle feedlot manure composting. *Journal of Environmental Quality*, 33(1), 37–44. https://doi.org/10.2134/jeq2004.0037

Havlík, P., Valin, H., Herrero, M., Obersteiner, M., Schmid, E., Rufino, M. C., & Notenbaert, A. (2014). Climate change mitigation through livestock system transitions. *Proceedings of the National Academy of Sciences*, 111(10), 3709-3714.

Hazarika, J., & Khwairakpam, M. (2018). Evaluation of biodegradation feasibility through rotary drum composting recalcitrant primary paper mill sludge. *Waste Management*, 76, 275–283. https://doi.org/10.1016/j.wasman.2018.03.04

He, X., Yin, H., Sun, X., Han, L., & Huang, G. (2018). Effect of different particle-size biochar on methane emissions during pig manure/wheat straw aerobic composting: Insights into pore characterization and microbial mechanisms. *Bioresource Technology*, 268, 633–637. https://doi.org/10.1016/j.biortech.2018.08.047

He, X., Yin, H., Han, L., Cui, R., Fang, C., & Huang, G. (2019). Effects of biochar size and type on gaseous emissions during pig manure/wheat straw aerobic composting: Insights into multivariate-microscale characterization and microbial mechanism. *Bioresource Technology*, 271(C), 375–382. https://doi.org/10.1016/j.biortech.2018.09.104

Himanen, M., & Hänninen, K. (2011). Composting of bio-waste, aerobic and anaerobic sludges–effect of feedstock on the process and quality of compost. *Bioresource Technology*, 102(3), 2842-2852.

Hindman, E. E., & Upadhyay, B. P. (2002). Air pollution transport in the Himalayas of Nepal and Tibet during the 1995–1996 dry season. *Atmospheric Environment (1994)*, 36(4), 727–739. https://doi.org/10.1016/S1352-2310(01)00495-2

Hua, L., Wu, W., Liu, Y., McBride, M. B., & Chen, Y. (2009). Reduction of nitrogen loss and Cu and Zn mobility during sludge composting with bamboo charcoal amendment. *Environmental Science and Pollution Research International*, 16(1), 1–9.

Hubbe, M., Nazhad, M., & Sanchez, C. (2010). Composting as a way to convert cellulosic biomass and organic waste into high-value soil amendments: A review. *Bioresources*, 5(4), 2808–2854. https://doi.org/10.15376/biores.5.4.2808-2854

Hue, N. V., & Liu, J. (1995). Predicting compost stability. *Compost Science & Utilization*, 3(2), 8-15.

Horisawa, S. (1999). Biodegradation of non-lignocellulosic substances II: Physical and chemical properties of sawdust before and after use as artificial soil. *Journal of Wood Science*, 45(6), 492–497. https://doi.org/10.1007/BF00538959

IPCC (2007). Climate change 2007: mitigation of climate change: contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. https://www.ipcc.ch/site/assets/uploads/2018/03/ar4_wg3_full_report-1.pdf

Jørgensen, K., & Jensen, L. S. (2009). Chemical and biochemical variation in animal manure solids separated using different commercial separation technologies. *Bioresource Technology*, 100(12), 3088–3096. https://doi.org/10.1016/j.biortech.2009.01.065

Jiang, T., Schuchardt, F., Li, G. X., Guo, R., & Luo, Y. M. (2013). Gaseous emission during the composting of pig feces from Chinese Ganqinfen system. *Chemosphere* (Oxford), *90*(4), 1545–1551. https://doi.org/10.1016/j.chemosphere.2012.08.056

Jiang, T., Ma, X., Tang, Q., Yang, J., Li, G., & Schuchardt, F. (2016). Combined use of nitrification inhibitor and struvite crystallization to reduce the NH_3 and N_2O emissions during composting. *Bioresource Technology*, 217, 210–218. https://doi.org/10.1016/j.biortech.2016.01.089

Joo, S. H., Monaco, F. D., Antmann, E., & Chorath, P. (2015). Sustainable approaches for minimizing biosolids production and maximizing reuse options in sludge management: A review. *Environmental Management*, 158, 133-145.

Karak, T., Bhattacharyya, P., Paul, R. K., Das, T., & Saha, S. K. (2013). Evaluation of composts from agricultural wastes with fishpond sediment as bulking agent to improve compost quality. *Clean: Soil, Air, Water*, *41*(7), 711–723. https://doi.org/10.1002/clen.201200142

Keener, H. M., Dick, W. A., & Hoitink, H. A. J. (2000). Composting and beneficial utilization of composted by product materials. *Land Application of Agricultural, Industrial, and Municipal By-products*. 6, 315-341.

Koivula, N., (2004). 05/01933 Ash in composting of source-separated catering waste. *Bioresource Technology*, 93, (3), 291–299. https://doi.org/10.1016/S0140-6701(05)81940-X

Kuba, T., Tschöll, A., Partl, C., Meyer, K., & Insam, H. (2008). Wood ash admixture to organic wastes improves compost and its performance. *Agriculture, Ecosystems & Environment*, 127(1), 43–49. https://doi.org/10.1016/j.agee.2008.02.012

Kulesza, S., Burns, J., Woodley, A., Gatiboni, L., Shupe, M., & Hicks, K. (2022). Distribution and fractionation of zinc and copper in poultry litters across North Carolina. *Communications in Soil Science and Plant Analysis*, 53(20), 2685–2699. https://doi.org/10.1080/00103624.2022.2072866

Kulikowska, D., & Bernat, K. (2022). Composting of municipal sewage sludge and lignocellulosic waste: Nitrogen transformations and humic substances molecular weight. *Energies* (Basel, Switzerland*)*, 16(1), 376–. https://doi.org/10.3390/en16010376

Kweku, D. W., Bismark, O., Maxwell, A., Desmond, K. A., Danso, K. B., Oti-Mensah, E. A., & Adormaa, B. B. (2018). Greenhouse effect: greenhouse gases and their impact on global warming. *Scientific Research and Reports*, 17(6), 1-9.

Larsen, K. L., & McCartney, D. M. (2000). Effect of C: N ratio on microbial activity and N retention: Bench-scale study using pulp and paper biosolids. *Compost Science & Utilization*, 8(2), 147-159.

Lehmann, J., & Joseph, S. (2010). Biochar for environmental management. *Science and Technology*. 2, 1-13.

Li, Y., Jin, P., Liu, T., Lv, J., & Jiang, J. (2019). A novel method for sewage sludge composting using bamboo charcoal as a separating material. *Environmental Science and Pollution Research International*, 26(33), 33870–33881. https://doi.org/10.1007/s11356-018-2534-9

Liang, Y., Leonard, J. J., Feddes, J. J. R., & McGill, W. B. (2006). Influence of carbon and buffer amendment on ammonia volatilization in composting. *Bioresource Technology*, 97(5), 748–761. https://doi.org/10.1016/j.biortech.2005.03.041

Lin, X., Wang, N., Li, F., Yan, B., Pan, J., Jiang, S., Peng, H., Chen, A., Wu, G., Zhang, J., Zhang, L., Huang, H., & Luo, L. (2022). Evaluation of the synergistic effects of biochar and biogas residue on CO_2 and CH_4 emission, functional genes, and enzyme activity during straw composting. *Bioresource Technology*, 360, 127608–127608.

López-Cano, I., Roig, A., Cayuela, M. L., Alburquerque, J. A., & Sánchez-Monedero, M. A. (2016). Biochar improves N cycling during composting of olive mill wastes and sheep manure. *Waste Management* (Elmsford), 49, 553–559. https://doi.org/10.1016/j.wasman.2015.12.031

Luo, Y., Liang, J., Zeng, G., Chen, M., Mo, D., Li, G., & Zhang, D. (2018). Seed germination test for toxicity evaluation of compost: Its roles, problems, and prospects. *Waste Management* (Elmsford), 71, 109–114. https://doi.org/10.1016/j.wasman.2017.09.023

Liu, N., Zhou, J., Han, L., Ma, S., Sun, X., & Huang, G. (2017). Role and multi-scale characterization of bamboo biochar during poultry manure aerobic composting. *Bioresource Technology*, 241, 190-199.

Lo, K. V., Lau, A. K., & Liao, P. H. (1993). Composting of separated solid swine wastes. *Agricultural Engineering Research*, 54(4), 307-317.

Maheswaran, G., Krishnapillai, M., Churchill, D., & Galagedara, L. (2020). Fly-ash from a pulp and paper mill: A potential liming material for agricultural soils in Western Newfoundland. *Canadian Biosystems Engineering*, 61(1), 1–1.15. https://doi.org/10.7451/CBE.2019.61.1.9

Malińska, K., Zabochnicka-Świątek, M., & Dach, J. (2014). Effects of biochar amendment on ammonia emission during composting of sewage sludge. *Ecological Engineering*, 71, 474–478. https://doi.org/10.1016/j.ecoleng.2014.07.012

Maynard, D.G., Kalra, Y. P., and Crumbaugh, J. A. (2007). Nitrate and exchangeable ammonium nitrogen. *Soil Sampling and Methods of Analysis*. 2, 71–80.

Manish, B., Richa, G., & Archana, T. (2013). Implementation of bulking agents in composting: A review. *Bioremediation and Biodegradation*, 4(7). 205

Manirakiza, E., Ziadi, N., Hamel, C., Lévesque, V., Antoun, H., & Karam, A. (2021). Soil microbial community dynamics after co-application of biochar and paper mill biosolids. Applied Soil Ecology: a Section of Agriculture, *Ecosystems & Environment*, 165, 103960. https://doi.org/10.1016/j.apsoil.2021.103960

Marche, Schnitzer, M., Dinel, H., Paré, T., Champagne, P., Schulten, H.-R., & Facey, G. (2003). Chemical changes during composting of a paper mill sludge–hardwood sawdust mixture. *Geoderma*, 116(3), 345–356. https://doi.org/10.1016/S0016-7061(03)00108-3

Mao, H., Lv, Z., Sun, H., Li, R., Zhai, B., Wang, Z., & Zhou, L. (2018). Improvement of biochar and bacterial powder addition on gaseous emission and bacterial community in pig manure compost. *Bioresource Technology*, 258, 195-202.

Mao, H., Zhang, H., Fu, Q., Zhong, M., Li, R., Zhai, B., & Zhou, L. (2019). Effects of four additives in pig manure composting on greenhouse gas emission reduction and bacterial community change. *Bioresource Technology*, 292, 121896.

Meena, A.L., Karwal, M., Dutta, D., Mishra, R.P. (2021). Composting: phases and factors responsible for efficient and improved composting. *Agricultural Food*, 3, 10028.

Munoz, G., Kelling, K., Powell, J., & Speth, P. (2004). Comparison of estimates of first-year dairy manure nitrogen availability or recovery using nitrogen-15 and other techniques. *Environmental Quality*, 33(2), 719–727. https://doi.org/10.2134/jeq2004.0719

Nag, M., Shimaoka, T., & Komiya, T. (2016). Nitrous oxide production during nitrification from organic solid waste under temperature and oxygen conditions. *Environmental Technology,* 37(22), 2890–2897. https://doi.org/10.1080/09593330.2016.1168485

Nagumo, Y., Oyanagi, W., Murakami, K., Tanahashi, T., Katoh, S., Andoh, T., Tsuchida, T., & Kato, N. (2011). Estimate of methane emission potential of poultry manure compost and swine manure compost from amount of acid detergent soluble organism. *Soil Science and Plant Nutrition*, 82(5), 401–404. https://doi.org/10.20710/dojo.82.5_401

Neher, D. A., Hoitink, H. A., Biala, J., Rynk, R., & Black, G. (2022). Chapter 17 - Compost use for plant disease suppression. *In The Composting Handbook.* Elsevier, 847-878. https://doi.org/10.1016/B978-0-323-85602-7.00015-7

Newfoundland and Labrador, (2016). The way forward climate change. Government Newfoundland and Labrador, (2020, June). https://www.gov.nl.ca/ecc/occ/action-plans/

Nordahl, S. L., Devkota, J. P., Amirebrahimi, J., Smith, S. J., Breunig, H. M., Preble, C. V., Satchwell, A. J., Jin, L., Brown, N. J., Kirchstetter, T. W., & Scown, C. D. (2020). Life-cycle greenhouse gas emissions and human health trade-offs of organic waste management strategies. *Environmental Science & Technology,* 54(15), 9200–9209. https://doi.org/10.1021/acs.est.0c00364

Pare, T., Dinel, H., Schnitzer, M., & Dumontet, S. (1998). Transformations of carbon and nitrogen during composting of animal manure and shredded paper. *Biology and Fertility of Soils*, 26(3), 173–178. https://doi.org/10.1007/s003740050364

Radziemska, M., Vaverková, M. D., Adamcová, D., Brtnický, M., & Mazur, Z. (2019). Valorization of fish waste compost as a fertilizer for agricultural use. *Waste and Biomass Valorization*, 10, 2537-2545.

Ricardo, O. O. E., Fernando, M. R. L., & Patricia, T. L. (2014). Evaluation of the addition of wood ash to control the pH of substrates in municipal biowaste composting. *Ingeniería, Investigación y Tecnología*, 15(3), 469-478.

Saer, A., Lansing, S., Davitt, N. H., & Graves, R. E. (2013). Life cycle assessment of a food waste composting system: environmental impact hotspots. *Cleaner Production*, 52, 234–244. https://doi.org/10.1016/j.jclepro.2013.03.022

Sánchez-García, M., Alburquerque, J. A., Sánchez-Monedero, M. A., Roig, A., & Cayuela, M. L. (2015). Biochar accelerates organic matter degradation and enhances N mineralization during composting of poultry manure without a relevant impact on gas emissions. *Bioresource Technology*, 192, 272-279.

Sayara, T., Basheer-Salimia, R., Hawamde, F., & Sanchez, A. (2020). Recycling of Organic Wastes through Composting: Process Performance and Compost Application in Agriculture. *Agronomy (*Basel*)*, 10(11), 1838. https://doi.org/10.3390/agronomy10111838

Sayara, T., & Sánchez, A. (2021). Gaseous emissions from the composting process: Controlling parameters and strategies of mitigation. *Processes*, 9(10), 1844. https://doi.org/10.3390/pr9101844

Serafimova, E., Mladenov, M., Michailova, I., Pelovski, Y. (2011). Study on the characteristics of waste wood ash. *University of Chemical Technology and Metallurgy*, 46 (7), 31–34. https://www.researchgate.net/publication/368231328

Shweta, Kumar, R., Singh, B. L., & Deepshikha, V. (2010). Integrating microbial composting and vermicomposting for effective utilization of by-products of sugar cane-processing industries. *Bioremediation*, 14(3), 158–167. https://doi.org/10.1080/10889868.2010.495365

Stiles, W. (2017). Poultry manure management. (2021, June). https://www.researchgate.net/publication/321001761

Sikora, L. J., & Sowers, M. A. (1985). Effect of temperature control on the composting process. *American Society of Agronomy*, 14(3), 434-439.

Sikora, L. J. (1999). MSW Compost reduces nitrogen volatilization during dairy manure composting. *Compost Science & Utilization*, 7(4), 34–41. https://doi.org/10.1080/1065657X.1999.10701982

Someshwar, A. V. (1996). Wood and combination wood-fired boiler ash characterization. *Environmental Quality*, 25(5), 962-972.

Sullivan, D. M., Bary, A. I., Miller, R. O., & Brewer, L. J. (2018,). Interpreting Compost Analyses. OSU Extension Catalog, 1–10. https://catalog.extension.oregonstate.edu/sites/catalog/files/project/pdf/em9217.pdf

Sun, X., Lu, P., Jiang, T., Schuchardt, F., & Li, G. (2014). Influence of bulking agents on CH_4, N_2O, and NH_3 emissions during rapid composting of pig manure from the Chinese Ganqinfen system. *Zhejiang University. B. Science*, 15(4), 353–364. https://doi.org/10.1631/jzus.B13a0271

Sun, Z. Y., Zhang, J., Zhong, X. Z., Tan, L., Tang, Y. Q., & Kida, K. (2016). Production of nitrate-rich compost from the solid fraction of dairy manure by a lab-scale composting system. *Waste Management*, 51, 55-64.

Sundberg, C., Smårs, S., & Jönsson, H. (2004). Low pH as an inhibiting factor in the transition from mesophilic to thermophilic phase in composting. *Bioresource Technology*, 95(2), 145-150.

Szanto, G., Hamelers, H. V., Rulkens, W., & Veeken, A. H. (2007). NH3, N2O and CH4 emissions during passively aerated composting of straw-rich pig manure. *Bioresource Technology*, 98(14), 2659–2670. https://doi.org/10.1016/j.biortech.2006.09.021

Tiquia, S. M., Tam, N. F. Y., & Hodgkiss, I. J. (1996). Effects of composting on phytotoxicity of spent pig-manure sawdust litter. *Environmental Pollution*, 93(3), 249–256. https://doi.org/10.1016/S0269-7491(96)00052-

Thambirajah, J. J., Zulkali, M. D., & Hashim, M. A. (1995). Microbiological and biochemical changes during the composting of oil palm empty-fruit-bunches: Effect of nitrogen supplementation on the substrate. *Bioresource Technology*, 52(2), 133-144.

Thyberg, K. L., & Tonjes, D. J. (2017). The environmental impacts of alternative food waste treatment technologies in the U.S. *Cleaner Production*, 158, 101–108. https://doi.org/10.1016/j.jclepro.2017.04.169

Thunjai, T., Boyd, C., & Boonyaratpalin, M. (2004). Bottom soil quality in tilapia ponds of different age in Thailand. *Aquaculture Research*, 35(7), 698–705. https://doi.org/10.1111/j.1365-2109.2004.01072.x

Tripepi, R. R., Zhang, X., & Campbell, A. G. (1996). Use of raw and composted paper sludge as a soil additive or mulch for cottonwood plants. *Compost Science & Utilization*, 4(2), 26-36.

Turner, T., Wheeler, R., & Oliver, I. W. (2022). Evaluating land application of pulp and paper mill sludge: A review. *Environmental Management*, 317, 115439-115439.

Tuomela, M., Vikman, M., Hatakka, A., & Itävaara, M. (2000). Biodegradation of lignin in a compost environment: a review. *Bioresource Technology*, 72(2), 169-183.

Rajkovich, S., Enders, A., Hanley, K., Hyland, C., Zimmerman, A. R., & Lehmann, J. (2012). Corn growth and nitrogen nutrition after additions of biochar with varying properties to a temperate soil. *Biology and Fertility of Soils*, 48, 271-284.

Wang, C., Lu, H., Dong, D., Deng, H., Strong, P. J., Wang, H., & Wu, W. (2013). Insight into the effects of biochar on manure composting: Evidence supporting the relationship between N_2O emission and denitrifying community. *Environmental Science & Technology*, 47(13), 7341–7349. https://doi.org/10.1021/es305293h

Wang, X., Zhao, Y., Wang, H., Zhao, X., Cui, H., & Wei, Z. (2017). Reducing nitrogen loss and phytotoxicity during beer vinasse composting with biochar addition. *Waste Management* (Elmsford), 61, 150–156. https://doi.org/10.1016/j.wasman.2016.12.024

Wang, Q., Awasthi, M. K., Ren, X., Zhao, J., Li, R., Wang, Z., Wang, M., Chen, H., & Zhang, Z. (2018). Combining biochar, zeolite and wood vinegar for composting of pig manure: The effect on greenhouse gas emission and nitrogen conservation. *Waste Management Elmsford*, 74, 221–230. https://doi.org/10.1016/j.wasman.2018.01.015

Wong, J. W. C., Fung, S. O., & Selvam, A. (2009). Coal fly ash and lime addition enhances the rate and efficiency of decomposition of food waste during composting. *Bioresource Technology*, 100(13), 3324–3331. https://doi.org/10.1016/j.biortech.2009.01.063

Wu, S., He, H., Inthapanya, X., Yang, C., Lu, L., Zeng, G., & Han, Z. (2017). Role of biochar on composting of organic wastes and remediation of contaminated soils: A review. *Environmental Science and Pollution Research,* 24, 16560-16577.

Xiao, R., Awasthi, M. K., Li, R., Park, J., Pensky, S. M., Wang, Q., Wang, J. J., & Zhang, Z. (2017). Recent developments in biochar utilization as an additive in organic solid waste composting: A review. *Bioresource Technology*, 246, 203–213. https://doi.org/10.1016/j.biortech.2017.07.090

Xiao, X., Chen, B., Chen, Z., Zhu, L., & Schnoor, J. L. (2018). Insight into multiple and multilevel structures of biochar and their potential environmental applications:A critical review. *Environmental Science & Technology,* 52(9), 5027-5047.

Yang, F., Li, G. X., Yang, Q. Y., & Luo, W. H. (2013). Effect of bulking agents on maturity and gaseous emissions during kitchen waste composting. *Chemosphere* (Oxford), 93(7), 1393–1399. https://doi.org/10.1016/j.chemosphere.2013.07.002

Yin, Y., Yang, C., Li, M., Zheng, Y., Ge, C., Gu, J., Li, H., Duan, M., Wang, X., & Chen, R. (2021). Research progress and prospects for using biochar to mitigate greenhouse gas emissions during composting: A review. *The Science of the Total Environment*, 798, 149294–149294. https://doi.org/10.1016/j.scitotenv.2021.149294

Yin, Y., Yang, C., Tang, J., Gu, J., Li, H., Duan, M., Wang, X., & Chen, R. (2021). Bamboo charcoal enhances cellulase and urease activities during chicken manure composting: Roles of the bacterial community and metabolic functions. *Environmental Sciences* (China), 108, 84–95. https://doi.org/10.1016/j.jes.2021.02.007

Yuan, J. H., & Xu, R.-K. (2011). The amelioration effects of low temperature biochar generated from nine crop residues on an acidic Ultisol: Soil acidity corrected by biochar. *Soil Use and Management*, 27(1), 110–115. https://doi.org/10.1111/j.1475-2743.2010.00317.x

Yuan, H.R., Lu, T., Huang, H.Y., Zhao, D.D., Kobayashi, N., & Chen, Y. (2015). Influence of pyrolysis temperature on physical and chemical properties of biochar made from sewage sludge. *Analytical and Applied Pyrolysis*, 112, 284–289.

Zhang, L., & Sun, X. (2014). Changes in physical, chemical, and microbiological properties during the two-stage co-composting of green waste with spent mushroom compost and biochar. *Bioresource Technology,* 171, 274-284.

Zhao, R., Jiang, D., Coles, N., & Wu, J. (2015). Effects of biochar on the acidity of loamy clay soil under different incubation conditions. *Soils Sediments*, 15(9), 1919–1926.

Zucconi, F. (1981). Evaluating toxicity of immature compost. *Biocycle*, 54-57.