The impact of biobased residues on soil health and greenhouse gas emissions under a changing climate

by

Emmanuel Adedamola Badewa

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Examining Committee Membership

The following served on the Examining Committee for this thesis. The decision of the Examining Committee is by majority vote.

External Examiner: Kate Congreves
Associate Professor, Department of Plant Sciences,
University of Saskatchewan

Supervisor: Maren Oelbermann
Professor, School of Environment, Resources,
and Sustainability, University of Waterloo

Internal Member: Stephen Murphy
Professor, School of Environment, Resources,
and Sustainability, University of Waterloo

Internal-external Member: Fereidoun Rezanezhad
Research Associate Professor, Department of Earth and
Environmental Sciences, University of Waterloo

Other Member(s) Gordon Price
Professor, Department of Engineering,
Dalhousie University

Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

The dissertation is manuscript-based, with some material resulting from a joint research effort.
I am the first author of all the manuscripts and am mainly responsible for the preparation, development, data analysis, and collection found in this dissertation. The contributions from co-authors are as follows:

Chapter 1: A general introduction outlining the research background, brief experimental method, hypotheses, and objectives edited by Dr. Maren Oelbermann. The original experimental design is a joint effort by Dr. Maren Oelbermann and Dr. Joann Whalen (McGill University).

Chapter 2:
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Dr. Oelbermann reviewed the final manuscript and provided editorial advice and guidance.

Chapter 3:
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Agricultural soil with biobased residues can reduce annual greenhouse gas emissions.

Chun Chung Yeung provided the biobased residues information in the methodology. Dr. Oelbermann edited the chapter.

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Dr. Oelbermann provided ongoing guidance and supervision on the project design and data collection. All co-authors reviewed the final manuscript and provided editorial advice and guidance.

Chapter 5:
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Impact of biobased residues on soil health in a temperate agricultural field.

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Chapter 6:
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The Century Model evaluates the long-term effects of biobased residues on soil organic carbon dynamics.

Dr. Oelbermann edited the chapter.

Chapter 7:
This chapter covers the overall conclusion, including key research findings and contributions.
Dr. Oelbermann edited the chapter.

Abstract

The disposal of organic wastes in landfills and applying nitrogen fertilizers to agricultural land cause multiple environmental issues. However, recycling organic wastes, e.g., biobased residues for agricultural use, is increasingly adopted to address landfill waste disposal and excessive nitrogen fertilizer use.

Objectives: The overall aim of this thesis was to explore the impact of biobased residues on soil health and greenhouse gas emissions under a changing climate - to provide alternative fertilizer sources for farmers to address soil health and reduce nitrogen fertilizer use (Chapter 1). The objective of Chapter 2 was to explore and integrate the conceptual and theoretical issues necessary for the biobased residues approach in addressing soil security concerns under a changing climate. The objective of Chapter 3 was to quantify the annual greenhouse gas emissions of a silt loam soil amended with nitrogen fertilizer and biobased residues under corn-soybean rotation. Chapter 4 builds upon Chapter 3 and explicitly evaluates the greenhouse gas emissions during the spring freeze-thaw events. Chapter 5 considers biobased residues' capacity to improve soil health in a temperate agricultural field in Ontario, Canada. Chapter 6 demonstrated how biobased residues, compared to nitrogen fertilizer, affect soil organic carbon stock and its associated fractions (active, slow, and passive) under continuous cropping and crop rotation in Ontario, Canada, using the Century model.

Methods: Chapter 2 is a literature review focused on establishing the links and assessment factors between sustainable indicators and biobased residues for soil security. Chapter 3 determined greenhouse gas emissions of soil amended with biobased residues (compost, biosolids, digestate) and nitrogen fertilizer under corn-soybean rotation in Elora Research Station, Ontario. Chapter 4 quantified greenhouse gas emissions, specifically during the spring freeze-thaw, where the soil was categorized as waterlogged, wet, or dry. Chapter 5 focuses on soil sampling and crop harvest carried out in autumn of each field season at a field site located in Elora, Ontario, Canada. Chapter 6 focuses on historical and current agroecosystem management practices in Elora, Ontario. This involved using the Century Soil Organic Matter model to predict future soil organic carbon stock changes using eight agroecosystem management practices.

Results and Discussion: Chapter 2 identified that biobased residues could address the underlying waste and agricultural issues, and more research on biobased residues dynamics in agricultural soil is critical. Chapter 3 demonstrated that biobased residues have a lower non-carbon dioxide (nitrous oxide and methane) emission than nitrogen fertilizer during the non-growing season. Chapter 4 showed that the dry phase during freeze-thaw, due to enhanced warming, caused intensified carbon dioxide flux compared to the wet and waterlogged freeze-thaw phase. Chapter 5 revealed that the soil health score of biosolids, nitrogen fertilizer, and all treatments combined contributed to one or more components of crop productivity. Chapter 6 demonstrated that agroecosystem management practices with compost and biosolids improved soil organic carbon's long-term (150 years) stabilization.

Overall Conclusions: biobased residues can function as an alternative for nitrogen fertilizer since they have the potential to mitigate greenhouse gas emissions, improve soil health, and increase the long-term stability of soil organic carbon that leads to carbon sequestration (Chapter 7).

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Dedication

This dissertation is dedicated to God almighty giver of life, grace, and strength. To all who have contributed and supported me. You all made it possible.

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Chapter 1
Introduction

1.1 Problem Context and Research Rationale

The world is faced with the challenge of addressing food and soil security, soil health, and environmental issues such as waste, pollution, and greenhouse gas emissions. The continuously growing global population has led to an increase in the disposal of organic wastes into landfills (Ho et al., 2017). Organic wastes such as food waste, leaf and yard waste, and wastewater biosolids are associated with multiple environmental issues (Jacobs and McCreary, 2003, Muniyasamy, 2016). Approximately, 50% of the food produced worldwide is wasted and the storage of this organic wastes requires large tracts of land that are potentially useful for food, fuel, feed, or fibre production (Segrè and Gaiani, 2012). The world produced 2 billion metric tonnes of solid waste per year, out of which United states produced 12% that filled and covered ~12000 hectares of land (EPA, 2022). Furthermore, the decomposition of organic wastes contributes to air and water contamination and greenhouse gas emissions (Faubert et al., 2019). In addition, the high quantity of nitrogen fertilizers used by conventional agriculture results in numerous environmental problems such as soil degradation and greenhouse gas emissions (Altieri and Nicholls, 2012, Li and Chen, 2020; Shakoor et al., 2021).

Agricultural land application of organic waste can serve as an alternative fertilizer source to address the issues of food security, soil health, and environmental degradation.

Nitrogen fertilizers ignore the health of the soil and have contributed more to greenhouse gas emissions than organic amendments (Bruges, 2010; Toor et al., 2021). Additionally, the production of nitrogen fertilizers has a high greenhouse gas footprint compared to organic amendments (Hathaway, 2016, Roman-Perez et al., 2021). The disposal of organic wastes into landfills also contributes to the emission of nitrous oxide (N_2O) and methane (CH_4), potent greenhouse gases with a global warming potential 273 and 28 times greater than that of carbon dioxide (CO_2) (Levis and Barlaz, 2011; IPCC, 2021). However, organic wastes can be diverted and processed to produce organic matter and nutrient-rich soil amendments referred to as biobased residues that can enhance soil health and crop productivity, therefore playing a significant role in agricultural markets (Gauthier et al., 2011; Paustian et al., 2016; Chowdhury et al., 2017). The term "Biobased residues" is considered as organic amendments that have passed through biobased technological chains that help address negative reports and concerns of contamination preventing the adoption of organic wastes as amendments. The biobased residues, or organic amendments, are generated from organic wastes that help reduce the need to apply nitrogen fertilizers to agricultural land (Thangarajan et al., 2013). Furthermore, organic amendments of different sources and types, such as biosolids, composted food waste, and anaerobic digestate, are increasingly adopted for agricultural use (Stewart-Wade, 2020).

Understanding the impact of various biobased residues on soil health under a changing climate for agricultural use is critical. Soil health can be defined as the capacity of soils to provide a sink for carbon to mitigate climate change and a reservoir for storing essential nutrients for sustained ecosystem productivity (Toor et al., 2021). For example, biosolids contain high quantities of organic carbon, nitrogen, phosphorus, and other micronutrients, including sulfur, calcium, and iron – all of which are useful for plant growth (Wang et al., 2008; Sharma et al., 2017). Adding biosolids to agricultural and forest soils increases soil organic matter and water holding capacity, decreasing bulk density (Jin et al., 2015; Ouimet et al., 2015). Lal (2004a) found that under a changing climate, biosolids applied to agricultural soils increased resilience to drought due to enhanced water holding capacity. Recent studies showed that composted municipal organic waste positively influenced soil health, soil physical characteristics such as aggregate stability (Drury et al., 2014; Abujabhah et al., 2016), and decreased carbon dioxide emission (Paustian et al., 2016). A review by Nkoa (2014) found that anaerobic digestate positively affected soil physical properties such as bulk density and water holding capacity, increased nitrogen use efficiency, and reduced pollution and losses.

Furthermore, with increased climate variability, the intensity and severity of freeze-thaw events in temperate agricultural regions will become more frequent, resulting in greater greenhouse gas emissions, especially nitrous oxide emissions (Henry, 2007, 2013). Wagner-Riddle et al. (2017) further suggested that current studies do not consider nitrous oxide emission during freeze-thaw events, leaving the possibility of underestimating this greenhouse gas emission by 28%. Most agricultural research also ignores non-growing season events and amendment applications that strongly drive greenhouse gas emissions (Adair et al., 2019). Hence, it is critical to understand the impact of biobased residues under the changing climate and seasons in agricultural land.

1.1 Research Objectives

The research objectives of this dissertation were to explore the impact of biobased residues on soil health and greenhouse gas emissions under a changing climate. The aim was to evaluate if biobased residues could play a significant role as an alternative source of fertilizer while improving soil health and reducing greenhouse gas emissions compared to nitrogen fertilizer.

1.1.1 Chapter 2

Achieving Soil Security Through Biobased Residues: The objective of this chapter was to explore and integrate the conceptual and theoretical issues necessary for the biobased residues approach in addressing soil security concerns under a changing climate.

1.1.2 Chapter 3

Agricultural soil with biobased residues can reduce annual greenhouse gas emissions: The objectives of this chapter were to 1) quantify the annual greenhouse gas emissions of a silt loam soil amended with nitrogen fertilizer and biobased residues; 2) evaluate the non-growing season contribution to annual emissions; and 3) assess the key soil factors regulating the greenhouse gas fluxes in a temperate agricultural field.

1.1.3 Chapter 4

Spring freeze-thaw stimulates greenhouse gas emissions from agricultural soil: The objectives of this chapter were to quantify and compare greenhouse gas fluxes from soil amended with biobased residues and nitrogen fertilizer during the spring freeze-thaw events.

1.1.4 Chapter 5

Impact of biobased residues on soil health in a temperate agricultural field: The objectives of this chapter were to determine 1) whether biobased residues can improve soil health and crop productivity; 2) is there any correlation between the soil health (indicators) and crop yields?; and (3) to contribute to the database of soil health studies across varied soil types especially for Canada.

1.1.5 Chapter 6

The Century Model evaluates the long-term effects of biobased residues on soil organic carbon dynamics.: The goal was to assess how biobased residues, compared to nitrogen fertilizer, affect soil organic carbon stock and its associated fractions (active, slow, and passive) under continuous cropping and rotation in Elora, Ontario, Canada using the Century model.

1.2 Dissertation Structure

This dissertation is presented in manuscript format. Chapter 1 begins with the introduction, objectives of the dissertation, and the research papers presented in the following chapters. The body of this dissertation is five manuscripts that address the potential of biobased residues for soil health, crop productivity, and resilience to climatic extremes during the non-growing season.

Chapter 2 is a review of biobased residues in relation to soil security. This chapter was published as a peer-reviewed article in *World journal of Agriculture and Soil Science* in 2020. Chapter 3 evaluates the effects of biobased residues on greenhouse gas emissions, during temperate growing and non-growing seasons. Chapter 4 considers the impact of spring freeze-thaw events on greenhouse gas emissions from soil amended with biobased residues. This chapter was published as a peer-reviewed article in the *Journal of Frontiers in Environmental Science.* The formatting of these papers has been modified to adhere to the dissertation requirements. There have been no changes made to the content of the papers. Chapter 5 evaluates the impact of biobased residues on soil health. Chapter 6 assesses the capacity of biobased residues to contribute to the soil organic carbon in a temperate soil. Chapter 7 is the concluding chapter that integrates all the major findings and presents key contributions and recommendations for further research.

1.3 Research location field experimental design

The experimental field site was located at the Elora Research Station, Elora, Ontario (43°45'N, 80°21'W) herein referred to as Elora.

The climate is humid continental temperate with an annual mean temperature of 6.6°C and mean annual precipitation of 861 mm (Elora). The soil texture and soil type was a silt loam grey brown Luvisol. Soil characteristics before the experiment at Elora were 25 g C kg^{-1} and 2.4 g N kg^{-1} with a pH of 7.9.

The field experiment was initiated in May 2018 with a completely randomized block design with four treatments and four replicates per treatment. The treatments included 1) commercial granular nitrogen fertilizer (urea; 170 kg N ha^{-1}); 2) hydrolyzed biosolids slurry (biosolids; 28,000 l ha^{-1}); 3) composted food waste (compost; 12 t ha^{-1}); and 4) liquid anaerobic digestate of farm green plant (digestate; 42,000 l ha^{-1}) (Figure 1.1). The fertilizer and amendment application rates were based on standard agronomic practices and local agronomic requirements (Table 1.1). Biobased residues used in this research were based on cooperation with Industrial partners including Lystek (biosolids), AIM Environmental (compost) and BioEn (anaerobic digestate).

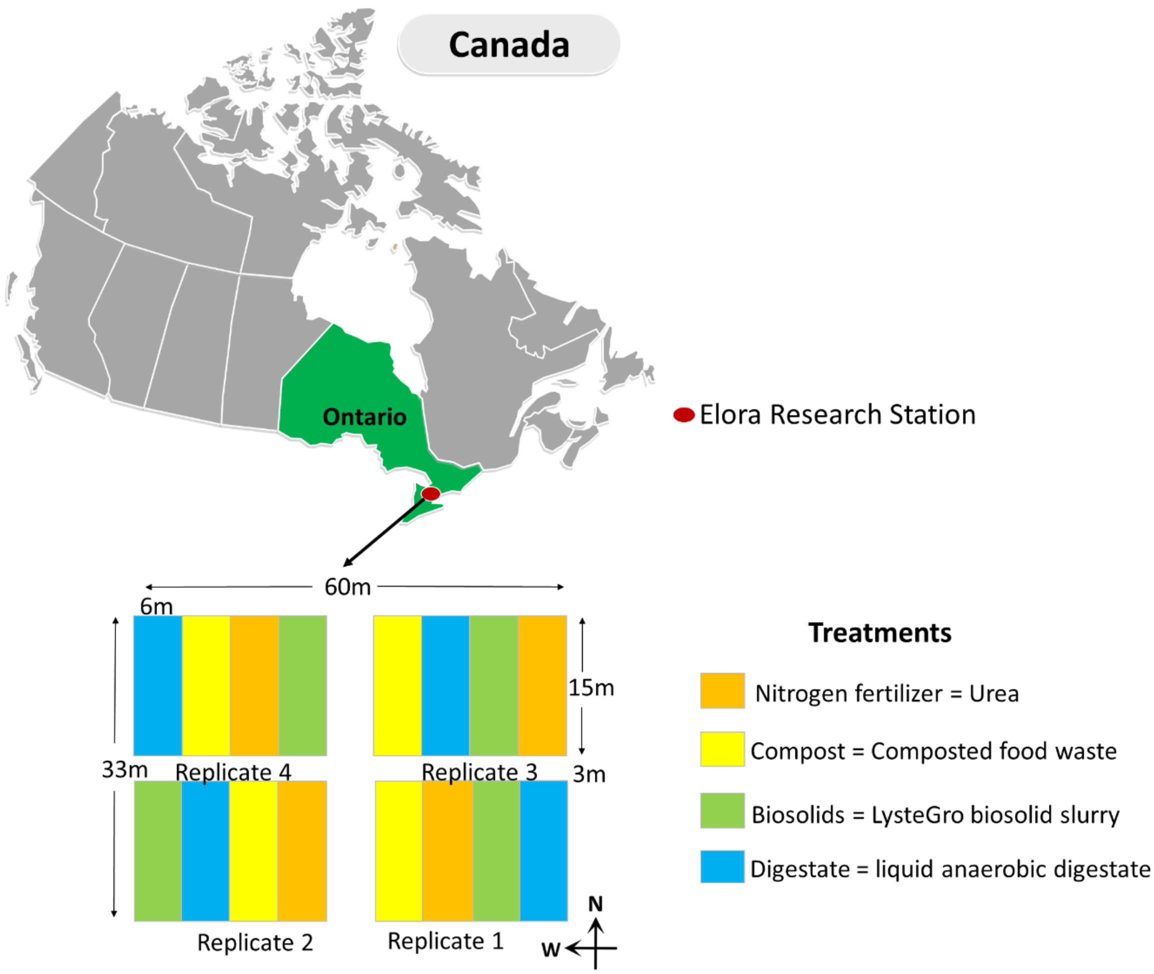


Figure 1.1: The location and experimental design of the field site at Elora Research Station, Ontario, Canada.

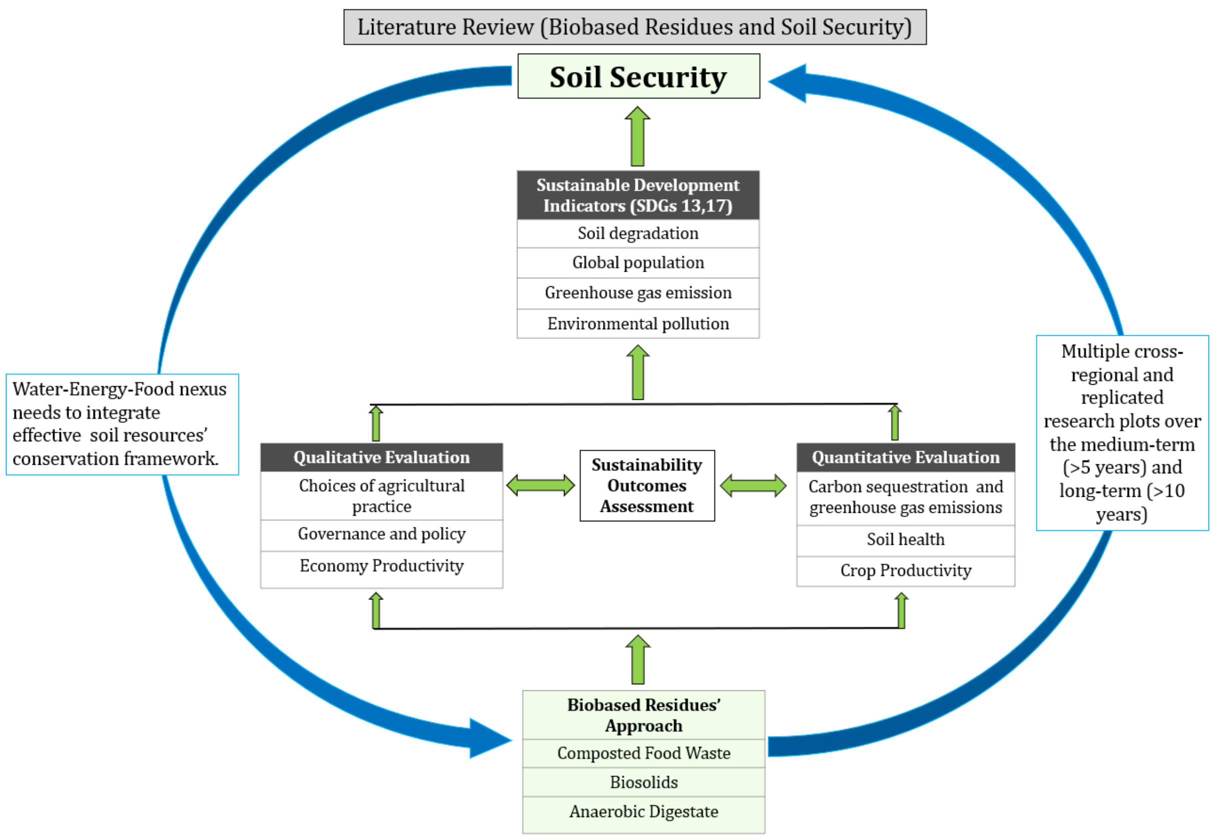
Table 1:1: Agronomic management practices during the experiment at Elora Research Station, Ontario, Canada.

Year	Date	Elora
2018	22 May	Prepare soil for seeding (disc harrow)
	24 May	All Organic fertilizer applications: Composted food waste: 12t/ha Biosolids: 28000L/ha Digestate: 42000L/ha
	25 May	NPK application (170 kg N/ha; 27 kg P_2O_5/ha; 55 kg K_2O/ha) Fertilizer incorporation Seeding corn (78,500 /ha of DeKalb DKC-3855RIB; 75 cm inter-row spacin)
	28 May	Herbicide application: Frontier Max (dimethenamid-p/Banvel (dicamba)/Atrazine tankmix (720/165/554 g active ingredient /ha)
	14 June	Herbicide application: Roundup Weathermax (glyphosate) 1350g/ha
	18 Oct	Corn harvest (grain only)
	1 Nov	Moldboard plowing 20 cm deep
2019	7 June	Field Cultivation (disc harrow)
	12 June	Seeding soybean (DKB 003-29, 450000 seeds per hectare, 15cm inter-row spacing)
	11 Oct	Soybean harvest (grain only), stalk residue chopped, and disc harrowed
2020	May 12	All Organic fertilizer applications: Composted food waste: 12t/ha Biosolids: 28000L/ha Digestate: 42000L/ha
	May 16	NPK application (170 kg N/ha; 27 kg P_2O_5/ha; 55 kg K_2O/ha) Fertilizer incorporation Seeding corn (78,500 /ha of DeKalb DKC-3855RIB; 75 cm inter-row spacin)
	May 22	Herbicide application: Frontier Max (dimethenamid-p /Banvel (dicamba)/Atrazine tankmix (720/165/554 g active ingredient /ha)
	Oct 14	All the grains, shoot, and roots were incorporated back into the soil
	March 24	Field experiment ended

Chapter 2

Achieving Soil Security Through Biobased Residues

Graphical Abstract



2.1 Introduction

The degradation of soil and the concurrent generation of organic waste is increasing as the world's population is equally on the rise. Annually, ~12 billion hectares of land are lost to soil erosion and degradation with an approximate cost of $2.3US trillion (USD) (Bouma, 2015). This startling rate of soil degradation and soil erosion has resulted in short- and long-term sustainability concerns for food, energy, and water (Nkonya et al., 2011; Lal, 2012; Delong et al., 2015). To address these challenges, the water-energy-food nexus was developed. The water-energy-food nexus addresses resource and development challenges that improve the understanding of the complex interactions among multiple resource systems (Foran, 2015; Wolfe et al., 2016; Albrechtet al., 2018). However, the adoption of the water-energy-food nexus has aggravated the degradation of soil (Hatfield et al., 2017), and there is no evidence that the water-energy-food nexus is sufficient to address concerns surrounding sustainability (Albrechtet al., 2018). Also, the water-energy-food nexus does not support future resources required by a rising global population, including the conservation of soil resources (Hatfield et al., 2017). However, healthy soil determines the capacity to produce food, fodder, fibre, and energy (Hatfield et al., 2017). Hence, there is a need to consider and explore ways to maintain or improve long-term soil health and soil security using sustainable approaches. Soil security is defined as the ability for soil to sustain functions to provide planetary services and human wellbeing while soil health is defined as the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans (Doran et al., 1996 Bouma, 2015).

In addition to the water-energy-food nexus, intensive agroecosystem management practices are another major cause of global soil degradation. Therefore, it is pertinent to adopt sustainable approaches to agroecosystem management to maintain soil, crop, and livestock productivity immediately and in the long-term. Conventional agroecosystem management practices include a reliance on agrochemicals and nitrogen fertilizers that have compromised water quality and led to soil acidification (Lal, 2010; Gomiero et al, 2011). Currently, 70% of global nitrous oxide emission, a greenhouse gas with 310 times greater warming capacity than carbon dioxide, is derived from agriculture's reliance on nitrogen-derived mineral fertilizers (Lal, 2012). Additionally, intensive tillage practices have led to soil erosion which is paralleled by a loss of soil organic matter (Lal, 2010; Gomiero et al., 2011). To date, ~40% of the global croplands are experiencing significant soil erosion and degradation (Reynolds et al., 2007; DeLong et al.,2015). Due to the severity of global soil degradation, the United Nations declared 2015 as the International Year of Soils (http://www.fao.org/soils-2015/en/) to discuss possible ways of reversing and mitigating the rapid degradation of soil (Gomiero et al., 2011; United Nations, 2013). Furthermore, the International Union of Soil Scientists declared the years from 2015 to 2024 as the international decade of soils (https://www.iuss.org/international-decade-of-soils/) as a continuation of the accomplishments achieved during the International Year of Soils in 2015. These actions resulted in the integration of soil into 13 of the 17 United Nations Sustainable Development Goals (Keesstra et al., 2016).

The current understanding of various soil management practices and how these affect soil processes remain limited (Lal, 2012). This is because soil is a complex ecosystem comprised of abiotic and biotic components that interact with each other, and this is further complicated by interactions across the soil-crop-atmosphere continuum. In addition, various approaches to agroecosystem management including the type of crop planted and how/if crops are rotated, the quantity and type of amendments added to the soil (*e.g.,* manure, nitrogen fertilizer), and residue management (*e.g.,* complete, partial or no crop residue removal, residue input from sources outside of the farm) further add to this complexity (Helming et al., 2018). However, the United Nations - Food and Agriculture Organization is promoting a new pathway to soil conservation based on sustainable agricultural intensification (Rockström et al., 2017). This approach integrates a high level of productivity with the maintenance of a wide range of soil processes, which are critical to help maintain soil health under the current and projected increase for demand in food, fibre, fodder, and fuel (Rockström et al., 2017). The key concept of sustainable agricultural intensification includes the efficient use of natural resources via ecological interactions in the soil-plant-atmosphere continuum (Tittonell, 2014).

The concept of sustainable agricultural intensification also promotes the integration of the bioeconomy using organic residues that have been diverted from waste management processes (e.g., landfills, sewage), and the forestry, fishery, and agricultural industries (Birner, 2018). The bioeconomy, therefore, uses biological knowledge plus the resources that directly or indirectly originate from plants, animals, or microorganisms for commercial and industrial purposes (Birner, 2018). The biological resources used in the bioeconomy are referred to as biobased residues that transferred from biobased production chains to agricultural land (Ho et al., 2017). Current approaches to agriculture already include, to some extent, the use of biobased residue as a soil amendment; although with variation of the level of integration (Gomiero et al., 2011; Lampkin et al., 2015a). Despite some potential limitations of biobased residues like biosolids, the application of biobased residues is currently promoted as a way forward to a more sustainable approach to agriculture with the potential to enhance soil health, biodiversity, and climate change mitigation via carbon sequestration (Lampkin et al., 2015b; Luo et al., 2018). For example, in the Canadian Province of Ontario, organic amendments like biosolids are provided at no cost to agricultural producers (Wessuc, 2018; OMAFRA, 2020). However, their application to agricultural soil has limitations, and application is not permitted on steeply (>9%) sloped land, frozen soil, or soil with moderate to slow permeability, it also requires a 100 m buffer zone between area of application and aquatic ecosystems, (Wessuc, 2018; OMAFRA, 2020). However, biosolids that have been treated and meet requirements set by the Federal Fertilizers Act (minimal heavy metal and pathogen content) can be sold as a fertilizer (e.g., LysteGro; https://lystek.com/solutions/lystegro-biofertilizer/) to agricultural producers (Halloran, 2020). Other biobased residues including composted food waste (compost), anaerobic digestate (digestate) as well as biosolids, have the capacity to meet the ecologically based agronomic and soil management criteria necessary to achieve soil security and sustainability. Composted food waste had the greatest positive long-term effect on crop yield compared to a fertilizer-only control (Drury et al., 2014). For example, Drury et al. (2014) found that compost increased yields by 11.3% compared to the fertilized control over a 10-year period. While biosolids contain high quantities of organic carbon, nitrogen, phosphorus, and other micronutrients including sulfur, calcium and iron (Wang et al., 2008), they also contain heavy metals and pathogens allowing their application only on a 5-year rotation (Quilty and Cattle, 2011). Anaerobic digestates improve soil physical characteristics (reduced erosion, improved soil structure and increased water retention) and enhance microbial activity (increased biomass and nitrogen mineralization rates) compared to soils amended with manure or nitrogen fertilizer (Odlare et al., 2008; Boldrin et al., 2009). For example, Odlare et al. (2008) found that over a 4-year investigation, digestate increased the nitrogen mineralization capacity and the proportion of active soil microorganisms compared to nitrogen fertilizer.

2.2 Biobased residues in relation to sustainable development

The United Nations developed 17 Sustainable Development Goals based on several sustainability development indicators. The sustainability development indicators that encapsulate biobased residues include, but not limited to, a growing global population, environmental pollution, overflowing landfills, excessive use of agrochemicals, soil degradation and climate change. Since 1987, the Brundtland Sustainability Report noted that sustainability development indicators can be used effectively to solve global challenges affecting world's resource systems (Brundtland, 1987). This has resulted in the development of sustainability development indicators that are relevant, easy to understand, reliable and based on accessible data (Sustainable Measures, 2010). In addition, there is a school of thought that soil should be the basic criteria of ecological design after which plants, animals and people should be upwardly considered (Orr, 2002). This suggests the need to assess the effect of human activity on living organisms and their environment through a critical evaluation of areas involving soil health, food security, technological advancement, globalization, economic growth, and the environment (Dryzek, 2013; Living Planet Report, 2014).

Using biobased residues as a soil amendment has the capacity to address the underlying issues related to humans, the waste they produce and its application to agricultural soil to ensure food security (Figure 2.1). For example, the waste produced by humans has led to world's current crisis of landfills reaching capacity in addition to leaching and methane emissions. However, the organic portion of materials deposited in landfills can be recycled directly to agricultural soil or indirectly by first undergoing industrial processes, thereby supporting the bioeconomy (Birner, 2018). Biobased residues, when used as an agricultural soil amendment, improves soil health, enhances crop productivity, and mitigates climate change via carbon sequestration and lowering greenhouse gas emissions. A co-benefit of this approach is its capability to ensure the long-term security of food through enhanced soil health which also generates resilience in agroecosystems when adapting to a changing climate.

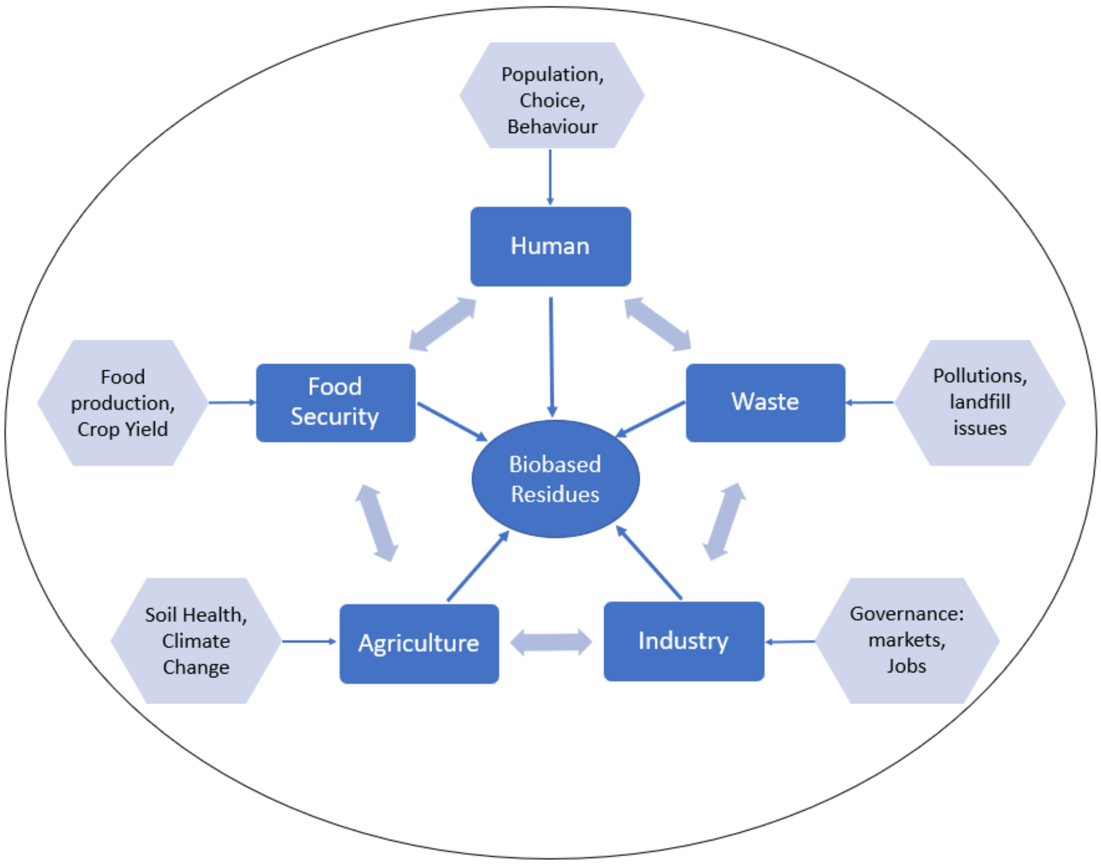


Figure 2.1: Conceptual model outlining the interconnections between sustainable indicators and biobased residues as a potential solution for soil security.

The success of this approach is strongly dependent on quantitative research at a regional scale to help determine how effective biobased residues are in their stability and efficacy in soil and their response to a changing climate. For example, do soils amended with biobased residue respond differently to freeze-thaw events in temperate biomes than soils amended with nitrogen fertilizer and/or livestock manure? The impact of biobased residue on GHG emissions during the growing season or under freeze-thaw has produced variable results (Thangarajan et al., 2013; Liu et al., 2016; Chantigny et al., 2019). There are only a few systematic studies that have evaluated the impact of biobased residues on the greenhouse gas balance, using a lifecycle approach, and soil health (Ho et al., 2017). Paustian et al. (2016) noted that the constraint to soil health is largely dependent on the amendment emission's lifecycle and the limitation of biobased residue application in cold climates. Furthermore, Urra et al. (2019) found that high application rates and odour causes a negative reaction towards biobased residue. Therefore, there is need to explore and carry out in depth investigations on biobased residues of various origins and how these affect soil health, carbon sequestration and greenhouse gas emissions under a changing climate. Knowledge mobilization, by integrating information gained by researchers and industry and its translation to agricultural produces and general society, will play a significant role in helping to understand that the integration of biobased residues to agroecosystems is a sustainable approach for the environment, people, and the economy.

2.3 The assessment of biobased residues for sustainability outcomes

2.3.1 Biobased residues: choice of agricultural practice and sustainability

The paradigm of sustainable intensification practices and the incorporation of BBRs within this model as a sustainable and complementary approach to agroecosystem management will help curb world's current reliance on nitrogen fertilizers (Lampkin et al., 2015b). Since the 1950s, the Green Revolution has served as a successful symbol of agricultural intensification, but it has also caused reliance on high-yielding hybrid crop varieties, irrigation infrastructure, use of agrochemicals such as herbicides and the reliance on nitrogen fertilizers to improve yield on the most impoverished soils (Borlaug, 1970). However, the Green Revolution has limitations because it relies on management approaches requiring materials (e.g., seed, fertilizer) that are not readily accessible for the majority of the global population (Borlaug, 1970; Altieri and Nicholls, 2012). The approach promoted by the Green Revolution is also not environmentally sustainable over the long-term due to the reliance on agrochemicals, hybridized crops, and the use of crops from outside of their native growing range (Borlaug, 1970; Altieri and Nicholls, 2012). Although, the understanding of biobased residues and the interactions they cause in agricultural soil remains limited (Ho et al., 2017), such knowledge is essential as it will determine the capacity of biobased residues to contribute to soil and food security on a global scale (Figure 2.2). In contrast to Green Revolution agriculture, agroecology where ecological processes are integrated into agricultural production systems, provides a sustainable approach to soil health and food security (Lampkin et al., 2015b; Gliessman, 2014) and it includes the use of biobased residue as part of routine agroecosystem management practices. Agricultural practices that incorporate biobased residue are also influenced by economic, social, and political factors (Garini et al., 2017). For example, Leopold (1970) notes that agricultural producers typically choose agroecosystem management practices that provide the greatest yield and therefore the highest economic gain. Thus, the rationale of agricultural producers, which is readily influenced by social and cultural factors, is critical, in incorporating biobased residue within already existing agroecosystem management practices (Hathaway, 2016). Additionally, if an agricultural producer views him or herself as a steward of the land, the choice will tend towards the adoption of sustainable agroecosystem management practices that often include biobased residue (Orr, 2002).

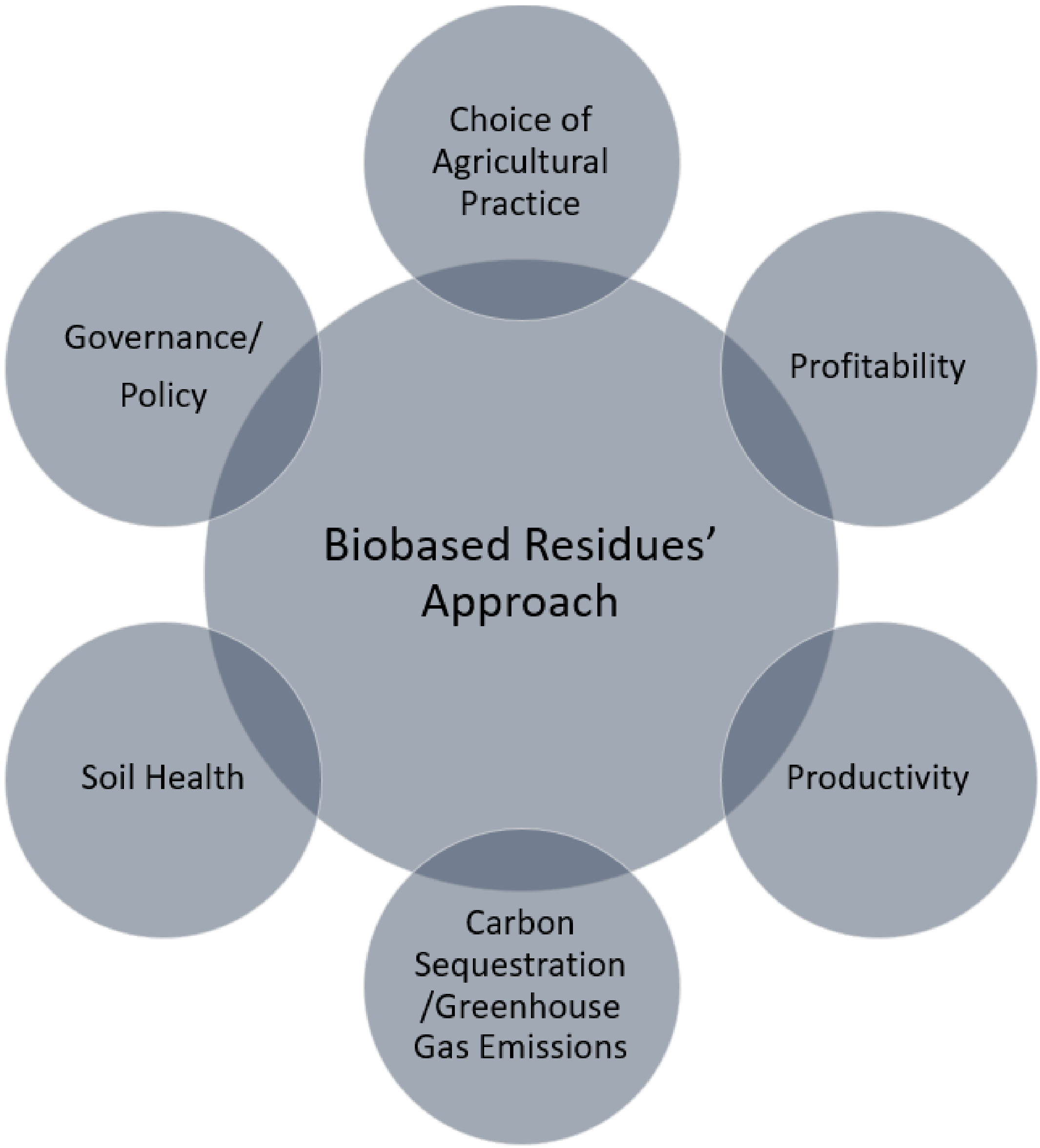


Figure 2.2: Conceptual model outlining the factors required to evaluate soil health and socioecological sustainability for biobased residues.

2.3.2 Governance and policy

Apart from the multiple agroeconomic and ecological benefits of biobased residue, there is a need for policy reforms that will support and recognize the role of biobased residue in maintaining soil health and food security. Various governmental and non-governmental organizations as well as industry have developed significant interest in integrating environmental waste incorporated with technology to enhance environmental sustainability including the mitigation of greenhouse gas (Amon et al., 2006; Saletnik et al., 2019; Adegbeye et al., 2020). Frequently, technological innovations that have incorporated biobased residue occur on a local scale using local feedstocks. Localized technology can be used to produce a nutrient rich soil amendment that can in turn improve soil health, enhance crop yield, and reduce greenhouse gas and non-greenhouse gas emissions including ammonia and dinitrogen (Meier et al., 2017; Yamashita et al., 2019; Adegbeye et al., 2020). Although the world is gradually gaining knowledge on the best approaches for sustainable agroecosystem management, this is not equally paralleled by advances in governance and policy. For example, de Molina (2013) noted that the movement of adopting sustainable agricultural practices stems largely from non-governmental organizations supported by academic institutions that are responsible for producing the required knowledge and technology. This means that there is a need for the development of agricultural policies that will motivate agricultural producers to adapt the use of biobased residue and for the general public to accept the use of biobased residue in the production of food, fodder, fibre and fuel. Garini et al. (2017) argued that "public policies are important because it can motivate the adoption of innovative farming practices". For instance, agricultural policies of the European Union acknowledge the complexity of socioecological systems and their dependence on sustainable agricultural practices to ensure food security (Lomba et al., 2017). This approach has also increased the value of agricultural land (Lomba et al., 2017). This infers that agricultural development and adoption of sustainable agroecosystem management practices, including the use of biobased residues, is dependent on the development and implementation of policies.

2.3.3 Carbon sequestration and greenhouse gas emissions

The use of organic manures in agriculture, based on crop and soil type, began more than 2000 years ago (Mang, 2015). Historically, the application of biobased residue was based on the need to recycle nutrients back into the soil without a conscious effort to maintain the soil organic matter level or sequester carbon. But instead, it was based on the premise of maintaining crop productivity. For example, organic manures such as human sewage and animal and plant residues were applied to China's agricultural soil to benefit crop growth (Beaton, 2009). In Medieval times, animal manure was applied on agricultural land to replace the materials removed by crop cultivation (Wild, 1988). By the end of the 18th Century, the use of organic amendments on agricultural soil began to shift and was nearly phased out by the 1950s when nitrogen fertilizers were introduced as a more effective way to increase crop productivity (Allison, 1944; Dyke, 1993; Goss et al., 2013). However, this approach has led to a steep decline in soil organic matter reserves while simultaneously increasing the accumulation of greenhouse gas in the atmosphere. This was based on the assumption that soil organic matter will always be available, and the accumulation of atmospheric greenhouse gas and climate change are extraneous (Lal, 2011; Altieri and Nicholls, 2012). Instead, the focus was on ensuring ample supply and availability of nitrogen since it is crucial in ensuring crop productivity (Lal, 2011; Altieri and Nicholls, 2012). However, the increasing accumulation of greenhouse gas in the atmosphere can be attributed to organic amendments and nitrogen fertilizers used in agricultural production systems (Hathaway, 2016; Yadav et al., 2019). To address these challenges, there are several efforts, such as the 4 per mille initiative (http://4p1000.org), whose focus is to increase soil organic matter content by 0.4% per year. This initiative targets long-term sequestration and storage of carbon in soil (Chenu et al., 2019) while simultaneously addressing climate change and helping improve soil fertility and crop productivity (Lal, 2008).

2.3.4 Soil health

Due to the importance of soil organic matter on soil processes that influence soil health, there has been a rising interest in understanding how amendment addition other than crop residues, manure, and fertilizer, influence soil health and enhance crop productivity. The use of amendments like biochar and biobased residue has the potential to enhance soil health and improve crop yield. Soil health, the capacity of soil to perform agricultural and environmental functions such as crop and biomass productivity (Lal, 2011), can be effectively assessed by evaluating physical, chemical, and biological soil characteristics referred to as soil health indicators (Allen et al., 2011). The most frequently evaluated soil health indicators include soil organic matter content, aggregate stability, microbial biomass and activity, soil carbon and nitrogen dynamics, and how the transformation of these nutrients relates to climate change (Lal, 2011). One of the main foci of soil health indicators is maintaining or enhancing soil organic matter levels (Altieri and Nicholls, 2012). Over the past 80 years, the role of soil organic matter as an ecosystem component and in maintaining soil health and agricultural productivity has been recognized. More recently, the role of soil organic matter in mitigating climate change due to its capacity to sequester carbon has also been realized (Schlesinger and Amundson, 2019). However, soil cultivation and agricultural production have been linked to declining reserves of soil organic matter, which has contributed 116 Pg of carbon to the atmosphere (Sanderman et al., 2017). In addition, the currently rapid expansion of agriculture in areas such as the Brazilian Amazon is causing a continual loss of soil organic matter and emission of carbon-based greenhouse gas into the atmosphere (Assad et al., 2013). Consequently, it is important to consider soil organic matter from a sustainable perspective and that this can be achieved using biobased residue. Integrating biobased residue with enhancing soil organic matter levels can also help with carbon trading or carbon offset policies (Murphy, 2015).

2.3.5 Productivity

The focus on increasing grain yield rather than maintaining soil health has led to a global concern that sustainable intensification should incorporate biobased residue into the modern crop production system. Due to the current and projected increase of the global population (Wise, 2013; Hatfield et al., 2017), cereal production must increase by 25-50%, from current production levels, by 2050 in order to generate sufficient food (Smith et al., 2010). However, the cost of increased agricultural production is soil degradation and, ultimately, its contribution to climate change (Tilman et al., 2011; Hatfield et al., 2017). An analysis by Wise (2013) also concluded that a large expanse of land is needed to increase crop production to support a growing global population. Therefore, there is a critical need to encourage sustainable agricultural intensification practices that include incorporating biobased residue to maintain soil health while ensuring crop productivity without further conversion of undisturbed ecosystems to agriculture. Hatfield et al. (2017) illustrated how the interaction of increased crop production is dependent on soil under sustainable intensification. In their paper, they also suggested improving and adapting agronomic techniques and increasing management intensity that recognizes soil resources management (Hatfield et al., 2017).

2.3.6 Profitability

One of the foundations of biobased residue is its capacity to contribute to the bioeconomy. Given that organic residues are readily available, and the projection of these waste materials will continue to increase and paralleled by an increase in the global population, there is a social and environmental need to recycle these materials (Quilty and Cattle, 2011; Maiti and Ahirwal, 2019). However, this also requires establishing infrastructure, including storage and co-composting facilities (Alvarenga et al., 2015; El-Naggar et al., 2019). Biobased residues are readily integrated into the bioeconomy, and their use can be expanded beyond agriculture to include the production of energy and other materials (Tsagaraki et al., 2017; European Commission, 2020). For example, the European bioeconomy in 2016 contributed €2.1 trillion (EUR) in addition to 18.3 million jobs, comprising ~9% of the total European Union workforce (BECOTEPS, 2011; Piotrowski et al., 2016). This implies that biobased residue can provide numerous benefits and at multiple scales that encourage sustainable development on a global scale.

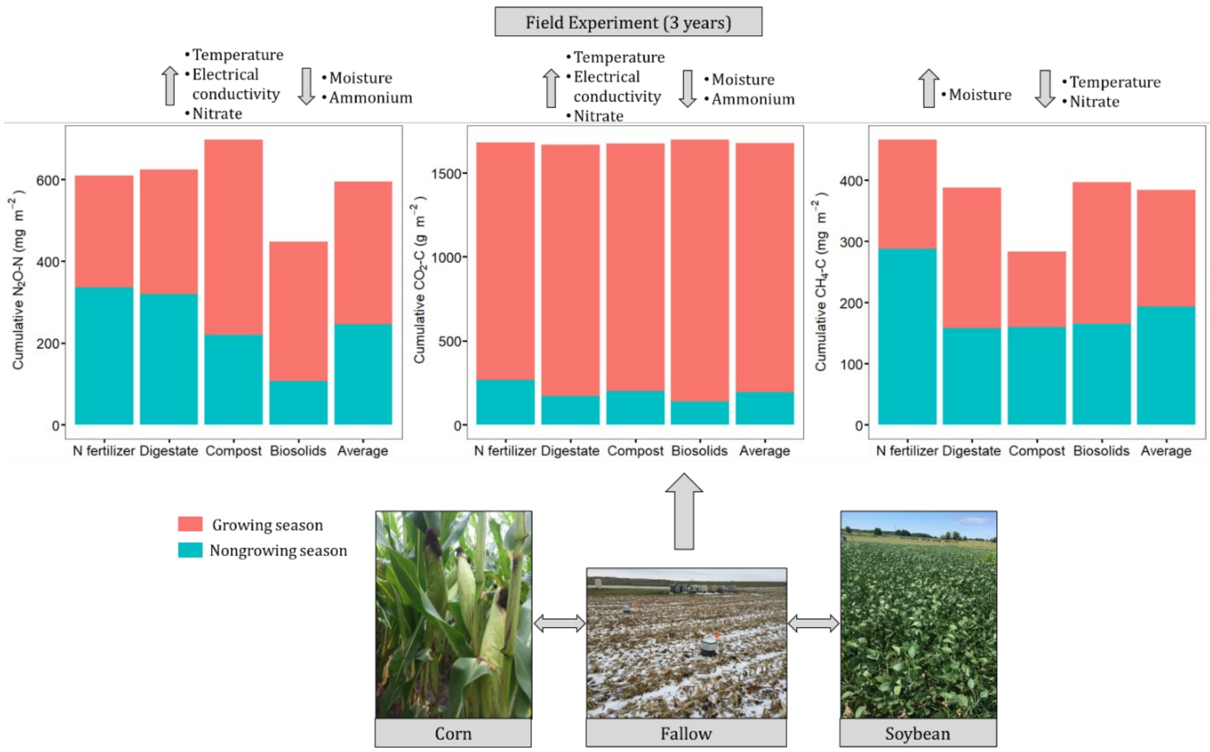
2.4 Conclusions and recommendations

The water-energy-food nexus is insufficient to address environmental sustainability issues since it does not integrate the conservation of soil resources into its framework. The importance of soil as part of a sustainable approach to agriculture is slowly gaining the attention of policy developers because of its role in ensuring food security. This acknowledgment is due to the international efforts by the United Nations - Food and Agriculture Organization that designated the year 2015 to the soil, the International Union of Soil Science that designated 2015-2024 as the decade of soils, and the 4 per mile movement initiated in France in 2015. These initiatives include the integration of biobased residue as a sustainable approach to agroecosystem management practices that help maintain soil health and ensure the long-term security of food. This mini-review outlined the relationship between biobased residue and sustainability development indicators and how this integration leads to a sustainable approach to agricultural land management practices. This chapter emphasizes the need for quantitative research on the impact of biobased residues on soil health, especially under a changing climate. To effectively explore the impact of integrating biobased residues into agricultural soil, it is pertinent to establish multiple cross-regional and replicated research plots over the medium-term (>5 years) and long-term (>10 years). This helps address modern agricultural and environmental challenges in a global bioeconomy.

Chapter 3

Agricultural soil with biobased residues can reduce annual greenhouse gas emissions.

Graphical Abstract



3.1 Introduction

Agriculture is responsible for 14% of global greenhouse gas emissions that consist of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Smith et al., 2014). The global warming potential of N_2O is ~ 273 times greater, and that of CH_4 is ~28 times greater than that of CO_2 (IPCC, 2021). Since the industrial revolution, emissions from agriculture, especially agricultural soils, have steadily increased due to world's reliance on nitrogen fertilizers (Lal, 2011; Weller et al., 2016). In addition, the form of nitrogen fertilizer applied to the soil strongly influences the amount of greenhouse gases emitted (Asgedom et al., 2014; Lazcano et al., 2016). Furthermore, limited available field measurements and nitrogen management practices different from inorganic nitrogen fertilizer, such as organic residues contribute, further contribute to the high disparity in quantifying greenhouse gas emissions (Wang et al., 2016; Yuan et al., 2017). Therefore, it is crucial to quantify long-term greenhouse gas emissions from agroecosystems to capture inter- and intraannual variability.

Greenhouse gas emissions from agricultural soils are regulated by biotic and abiotic factors and soil processes (Adair et al., 2019). For example, in contrast to CO_2 emissions produced from heterotrophic and autotrophic soil respiration (Van Zandvoort et al., 2017), CH_4 emissions are regulated by anaerobic microbial production (methanogenesis) or aerobic microbial consumption (methanotrophy) (Dutaur and Verchot, 2007; Kim et al., 2012). While, N_2O emissions result from nitrification or denitrification's nitrogen transformation processes (Charles et al., 2017). Furthermore, the controlling factors of greenhouse gas emissions include, but are not limited to, substrate (e.g., organic matter's quality and quantity, soil moisture content, nutrient content and availability, microbial activity, and soil temperature (Scott-Denton et al., 2006; Butterbach-Bahl, et al., 2013; Oertel, et al., 2016; Luan et al., 2019). However, adding nitrogen fertilizers and/or organic amendments such as biobased residues influence the different soil processes and factors that control the production or consumption of greenhouse gas (Rochette et al., 2008; Daly and Hernandez-Ramirez, 2020).

Seasonal variability of greenhouse gas emissions is uncertain due to a lack of data during the non-growing season (Wang et al., 2021). Most studies evaluating greenhouse gas emissions from agroecosystems have mostly focused on the growing season, with limited measurements taking place during the non-growing season (Adair et al., 2019). However, microbial respiration can persist during the non-growing season at soil temperatures <0°C (Miao et al., 2014; Natali et al., 2019) and therefore cause CO_2, N_20, and/or CH_4 emissions (Mastepanov et al., 2008; Merbold et al. 2013; Congreves et al., 2017). For example, Chantigny et al. (2016) found that CO_2 emission during the non-growing season was less than 20%, and N_2O emission was more than 50% of annual emissions. Treat et al. (2018) determined that CH_4 emissions were ~50% of the annual emission. Furthermore, Treat et al. (2018) also found that non-growing season emissions remain uncertain due to spatial and temporal variability. In addition, biobased residues could influence spatial and temporal emissions differently compared to nitrogen fertilizers. Consequently, it is critical to assess greenhouse gas emissions over the long term to understand the influence of temporal variability during the growing and non-growing (Chantigny et al., 2016; Wang et al., 2021).

The objectives of this chapter were to 1) quantify the annual greenhouse gas emissions of a silt loam soil, amended with nitrogen fertilizer and biobased residues; 2) evaluate the non-growing season contribution to annual greenhouse emissions; and 3) assess the key soil factors that control greenhouse gas emissions. It is hypothesized that i) biobased residues, compared to nitrogen fertilizer, will amplify greenhouse gas emissions due to their greater carbon and nitrogen content which boosts microbial activity; and ii) non-growing season net warming from N_2O and CH_4 emission will substantially contribute to annual greenhouse gas emission since temperate agricultural land is fallow and undergoes freezing during the non-growing season.

3.2 Materials and Methods

3.2.1 Experimental site and biobased residues

The experimental location was at Elora Research Station, Elora, Ontario, Canada. (43°45’N, 80°21’W, elevation 376m). The research was conducted from 2018 to 2021. The growing season was between May and October (crop seeding and harvesting), while the non-growing season was between November and April (post-harvest to spring thaw). The monthly mean temperatures ranged from 7^{o}C to 22^{o}C between May and September and -15^{o}C to 9^{o}C between October and April. The mean annual precipitation was 989 mm, based on a 16-year average from 2004-2019 (Environment Canada, 2021). The effect of ambient temperature, snow depth, and precipitation were obtained from a weather station located at the Elora Research Station (Figure 3.1). The source, feedstock substrate and method of production with chemical properties of the biobased residues used in the research were presented in Table 3.1.

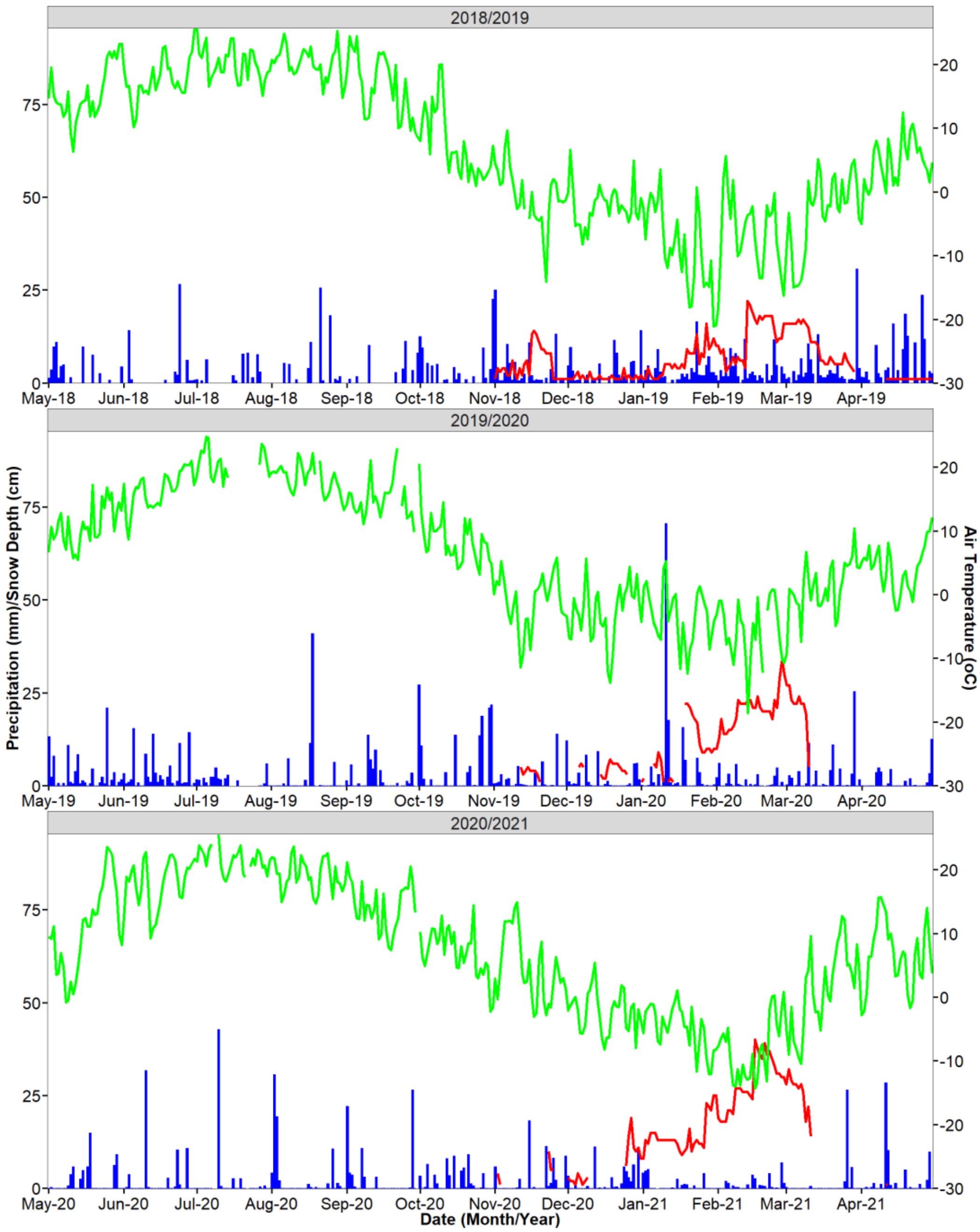


Figure 3.1: The variation in snow depth, precipitation, and soil and air temperature of the 2018 to 2021 growing and non-growing seasons at Elora Research Station, Ontario, Canada. Blue bars represent precipitation, the red line is the Snow depth, and the green line is the mean air temperature.

Table 3.1: Chemical properties of biobased residues used at Elora Research Station, Ontario, Canada.

Characteristics	Compost	Biosolids	Digestate
Supplier	AIM Environmental Group	Lystek International inc	BioEn Power inc
Feedstock	Food waste	Sludges	Green plant
Production process	Composting	Thermal hydrolysis	Anaerobic Digestation
Dry matter (%)	77.6	10.8	3.0
Total Nitrogen (%)	2.46	4.96	0.55
Total Phosphorus (%)	0.61	2.62	0.15
Total Potassium (%)	0.82	2.23	2.16
Calcium (%)	4.60	1.13	3.70
Magnesium (%)	0.59	0.08	0.42
Sodium (ppm)	7200	12170	26900
Iron (ppm)	3522	85634	18800
Manganese (ppm)	137.7	50.3	188
Copper (ppm)	56.3	520.41	65
Zinc (ppm)	126.2	575.2	290
% OM (w/w)*	48.1	6.25	0.23
% Water (w/w)	22.4	89.2	97.0
Total C/N	12	5.0	0.4

"%" for dry weight and ppm on a wet weight basis. "*" Percent weight relative to the wet weight of the applied fertilizer.

3.2.2. Experimental Design and management

The experimental field design at the field site at the Elora Research Station was a randomized complete block design (c.f., Chapter 1). Each treatment had four blocks (replicates) totaling 16 treatment replicate plots of size 6 x 12 m for each plot. Prior to the field set up, the location was under conventional practices of mixed soybean (*Glycine max L.),* canola (*Brassica rapa*), and barley (*Hordeum vulgare*). During the experiment, maize (*Zea mays* L.) was seeded in 2018 and 2020 and soybeans in 2019. After site cultivation, nitrogen fertilizer (urea) and biobased residues amendments (Table 3.2) were applied on May 24, 2018, and May 5, 2020, during the years maize was produced. The amendments were applied by surface broadcasting and incorporated with an offset disk harrow (0-10 cm). Crops were seeded in May of each year and harvested in October of each year. After harvest, the grains were removed, and the remaining crop residues were incorporated to a depth of 15 cm with an offset disc harrow. The site was fallow during the transient cold period from November to April.

Table 3.2: Total required fertilizers from biobased residues and nitrogen fertilizer during corn-soybean growing seasons at Elora Research Station, Ontario, Canada.

Treatment	Recommended application rate	Total N applied (kg ha^{-1})	% mineral-N (NH_4—N + NO_3^--N)	Total P applied (kg P_2O_5 ha^{-1})	Total K applied (kg K_2O ha^{-1})	Total organic Carbon applied (kg ha^{-1})
Nitrogen Fertilizer	170 kg ha^{-1}	170	100	27	55	-
Compost	12 t ha^{-1}	240	15.2	85.2	70.8	2880
Biosolids	28000 l ha^{-1}	215	47	260	116	1070
Digestate	42000 l ha^{-1}	231	97.5	60.5	75.5	92.4

Note: The recommended application rate for the biobased residues was based on the amount of N that will be relatively available in the first year of application (See Table S3.1 for N applied calculation). The fertilizer used for N was granular urea (46-0-0) applied at planting, P was triple superphosphate (0-46-0), and K was potassium chloride (0-0-60). P and K were broadcasted.

3.2.3. Greenhouse gas fluxes

Greenhouse gas fluxes were measured using closed static chambers constructed from polyvinyl chloride tubes. The chamber was 15 cm long and 56 cm in diameter and inserted into the soil to a 10 cm depth. The chamber volume was 30.2 L, with a 0.08 m^2 cross-sectional area. The chamber was randomly placed in each treatment replicate and removed only during planting and harvesting. Chamber caps were insulated and included a 1 cm diameter opening fitted with rubber septa for gas sampling and a 10 cm long (9 mm inner diameter) vent tube to account for pressure differences during sample collection (Parkin and Venterea, 2010). Caps were removed after each sampling event, and the soil was exposed to air to maintain equilibrium (Parkin and Venterea, 2010).

Gas samples were collected between 10:00-h and 16:00-h to account for diurnal temperature variation influence (Smith et al., 2003). Gas sampling (30 mL) was carried out at 0, 10, 20, and 30 min after placing the cap on the chamber and transferred into pre-evacuated 12 mL glass Exetainers (LabCorp Ltd., Lampeter, Wales, UK). Gases were analyzed on a chromatograph (Bruker Corporation, Billerica, MA, USA) with detection limits ≤ 100 ppm and a flame ionization detector (FID) set at 300 °C for CO_2 and CH_4 and an electron capture detector set at 350°C for N_2O. The gas concentration (ppm) was converted to mass-based concentration with the Ideal Gas Law using the relative molecular mass of carbon (12 g mol^{-1}) or nitrogen (28 g mol^{-1}) (Pedersen, 2017). The greenhouse gas flux was quantified using the HMR package in R (version 4.1.2), which integrated a linear and non-linear approach described by Hutchinson and Mosier (1981). Erroneous non-flux estimates from abnormal and low signal-to-noise ratios were returned as zero or incomplete fluxes (Pedersen et al., 2010).

Cumulative emissions were calculated using trapezoidal integration of the consecutive daily emissions (Van den Pol-van Dasselaar and Oenema, 1997; González-Méndez et al., 2020). The greenhouse gas fluxes with negative values were assumed to be zero to reduce the noise effect in the flux variation for trapezoidal integration (Levy et al., 2017). The yearly data collected were considered an adequate measure of the annual variability (González-Méndez et al., 2020). The global warming potential was calculated using equation (3.1) and (3.2).

$$CE = \sum F_i \text{ x } 24 \qquad (3.1)$$

where CE is the cumulative emission of soil greenhouse gas (μg m^{-2}/ mg m^{-2}) and Fi is the greenhouse gas flux of the i-th sampling time (μg m^{-2} h^{-1}). The GWP (g CO_2 eq m^{-2}) was calculated using:

$$GWP = CE_{CH_4} \text{ x } 27.9 + CE_{N_2O} \text{ x } 273 \qquad (3.2)$$

where, CE_{CH_4}, and CE_{N_2O} represent the cumulative emissions of CH_4, and N_2O, respectively.

Th CH_4 and N_2O for the season and annual cumulative emission expressed as CO_2 equivalents using a conversion factor of 27.9 for CH_4 and 273 for N_2O to determine the net warming effect for a 100-year time horizon (IPCC, 2021).

3.2.4. Soil parameters

At the same time as gas sampling, soil temperature, moisture, and electrical conductivity were measured 0-10 cm within a 1 m radius of each static chamber using a wet sensor (Delta-T Devices, Cambridge, United Kingdom) with detection limit 80% water content, ~600 mS m^{-1}, 40°C. Soil samples were also collected within 1 m radius to a 10 cm depth to quantify inorganic nitrogen (ammonium and nitrate) content. For ammonium and nitrate, 5 g of soil was extracted with 25 ml of 2.0M potassium chloride and analyzed colorimetrically on a UV-Vis spectrophotometer (Shimadzu UV-1800) at wavelengths 650 nm for ammonium and 540 nm for nitrate (Maynard and Kalra, 1993). Inorganic nitrogen stock (kg ha^{-1}) was determined by multiplying the nitrogen concentration (mg kg^{-1}), soil bulk density, and soil depth (Ruiz Diaz et al., 2008). Soil Inorganic nitrogen accumulation was fitted using a cumulative logistic distribution function package in R to estimate the maximum potential available nitrogen during the growing and non-growing seasons over the three field seasons.

3.2.5. Statistical analysis

Data processing and analysis were carried out using R *v.4.2.1* (R Core Team, 2020). Data were evaluated for normality and homogeneity of variance using Levene's and Shapiro-Wilk tests (Seltman, 2012). Parametric statistics was then adopted since data was found to be normal with equal variance and no violation of assumptions. The trapezoidal rule implemented through the *trapz* function in R was used for linear interpolation. Two-way factor analysis of variance was used to assess the differences among treatments, years, and their interactions for greenhouse gas fluxes and soil parameters. Differences among means were separated using Tukey's HSD multiple comparison test (Steel et al., 1997). The relationship between cumulative emissions and soil parameters during the growing and non-growing seasons was assessed using Pearson correlation coefficients (*r*) and a general additive model with the restricted maximum likelihood smoothing method. The correlation coefficient magnitude was interpreted as weak (0.10-0.39), moderate (0.40-0.69), and strong (0.70-0.89) (Schober et al., 2018).

3.3 Results

3.3.1 Greenhouse gas fluxes and soil parameters

The CO_2 flux had distinct seasonal pattern that was different among the years (p=0.001) but not different among treatments and their interactions (p>0.05) (Figure 3.2, Table S3.2). The CH_4 flux was different among years (p=0.001) but not among seasons and treatments (p>0.05) (Figure 3.2, Table S3.2). The N_2O flux was not significantly different among the years, seasons, and treatments (p>0.05) (Figure 3.2, Table S3.2). Soil moisture, temperature, and electrical conductivity were significantly different among seasons and years but not among the treatments and their interactions (p>0.05) (Figure 3.3, Table S3.2). Soil inorganic nitrogen (ammonium and nitrate) accumulation followed a non-linear pattern where year 1 nitrogen fertilizer treatment was significantly different from biobased residues ($p<0.05$). In contrast, year 2 and 3 were not different ($p>0.05$) (Figure 3.4).

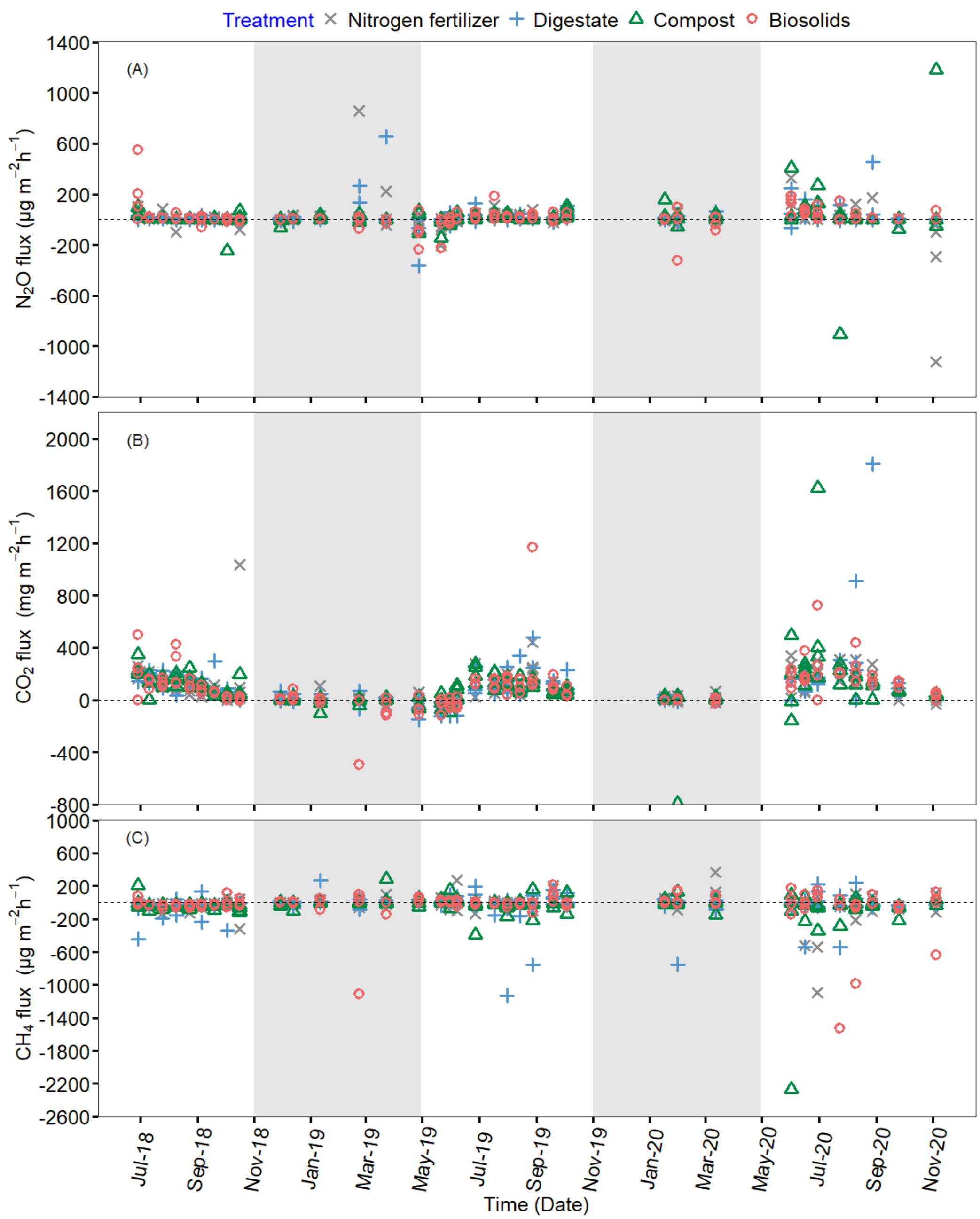


Figure 3.2: Greenhouse gas fluxes (a) nitrous oxide; N_2O (b) carbon dioxide; CO_2 (c) methane; CH_4 from the soil amended with nitrogen fertilizer, digestate, compost, and biosolids under growing and non-growing season at Elora Research Station, Ontario, Canada. The shaded area represents the non-growing season, and the broken line represents zero flux.

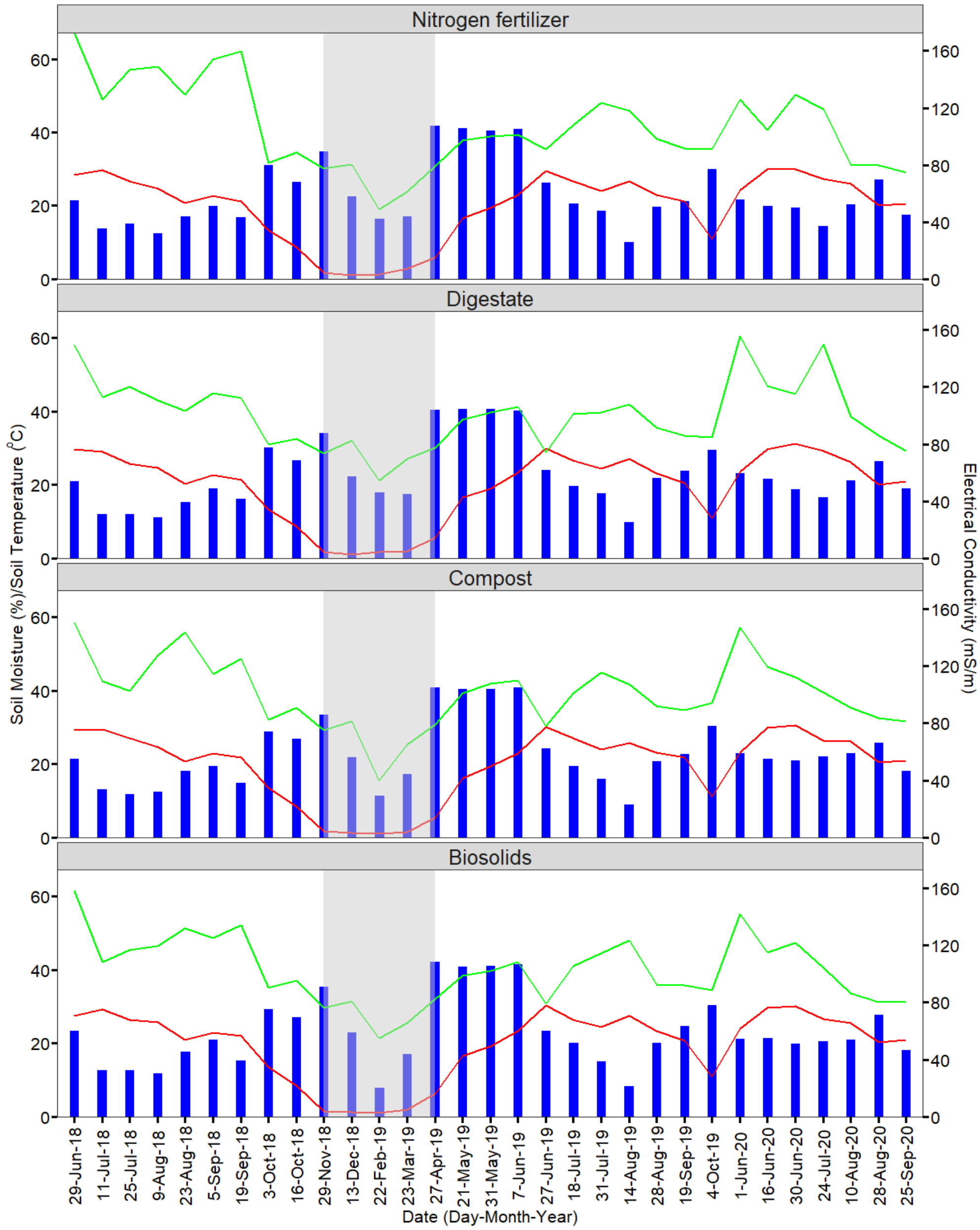


Figure 3.3: Soil moisture (blue bars), soil temperature (red line), and soil electrical conductivity (green line) under growing and non-growing seasons in treatments with nitrogen fertilizer, digestate, compost, and biosolids, at Elora Research Station, Ontario, Canada. The shaded area represents the non-growing season.

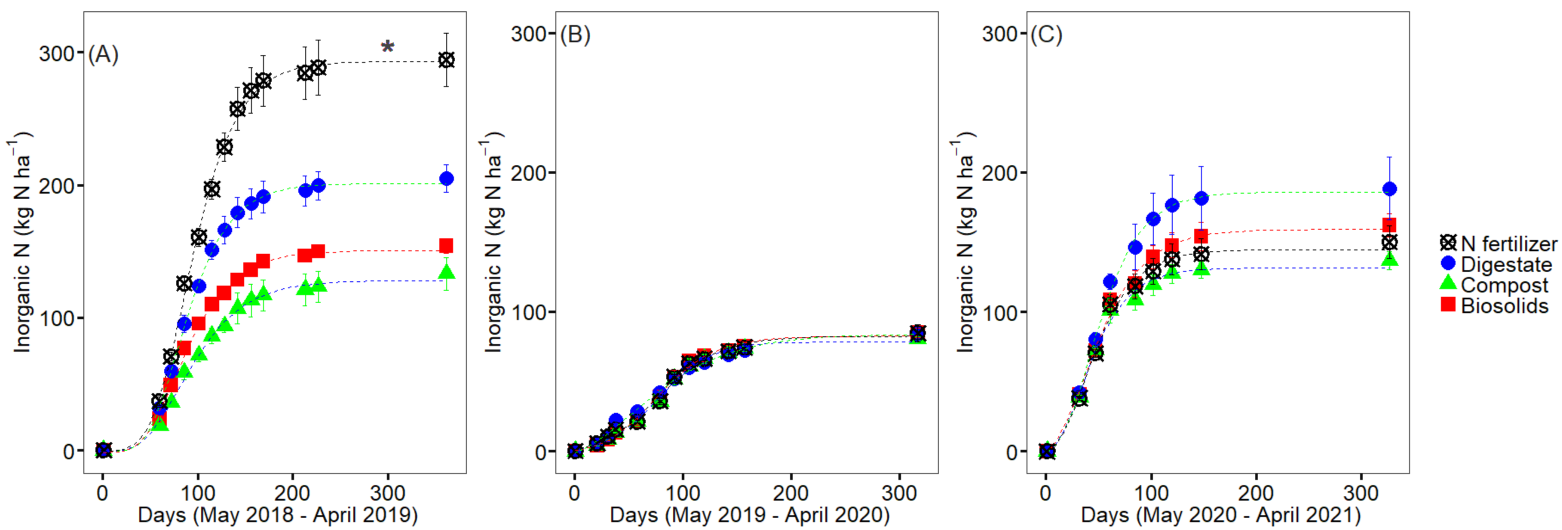


Figure 3.4: Soil inorganic nitrogen (ammonium and nitrate) accumulation from soil amended with biobased residues and nitrogen (N) fertilizer from (a) May 2018 - April 2019 (b) May 2019 - April 2020 (c) May 2020- April 2021 over the growing and non-growing season at Elora Research Station, Ontario, Canada. "*" represents significant differences (p<0.05) in nitrogen fertilizer treatment from biobased residues.

3.3.2 Treatment effects on cumulative emission and global warming potential

Overall, cumulative emissions and global warming potential were significantly different among treatments ($p<0.05$), except for the cumulative CO_2 emission (Figures 3.5 -3.8). Cumulative CH_4 emission and global warming potential were highest for nitrogen fertilizer and lowest for compost ($p<0.05$), while cumulative CO_2 emission was not significantly different among treatments. Cumulative N_2O emission was greatest for compost and lowest for biosolids ($p<0.05$). Cumulative CO_2 emissions from the nitrogen fertilizer treatment during the non-growing season accounted for 16% of the annual cumulative emissions (1680 g CO_2-C m^{-2}), while cumulative CH_4 emissions accounted for 62% of the annual cumulative emissions (465 mg CH_4-C m^{-2}) (Figures 3.5 and 3.6).

Compared to nitrogen fertilizer, biobased residues during the non-growing season reduced ($p<0.05$) CO_2 emissions by 10% for digestate, 12% for compost and 8% for biosolids, whereas CH_4 emissions decreased ($p<0.05$) by 41% (digestate), 56% (compost) and 42% (biosolids) (Figures 3.5 and 3.6). Cumulative N_2O emission from the nitrogen fertilizer treatment during the non-growing season (336 mg N_2O-N m^{-2}) accounted for 55% of the annual cumulative emissions (609 mg N_2O-N m^{-2}), while nitrogen fertilizer global warming potential in the non-growing season accounted for 61% of the annual global warming potential (114 g CO_2-eq m^{-2}) (Figures 3.7 and 3.8). Compared to nitrogen fertilizer, biobased residues during the non-growing season decreased ($p<0.05$) N_2O emissions by 51% (digestate), 32% (compost), and 24%

(biosolids), while the global warming potential by 41% for digestate, 56% for compost, and 42% for digestate (Figures 3.7 and 3.8).

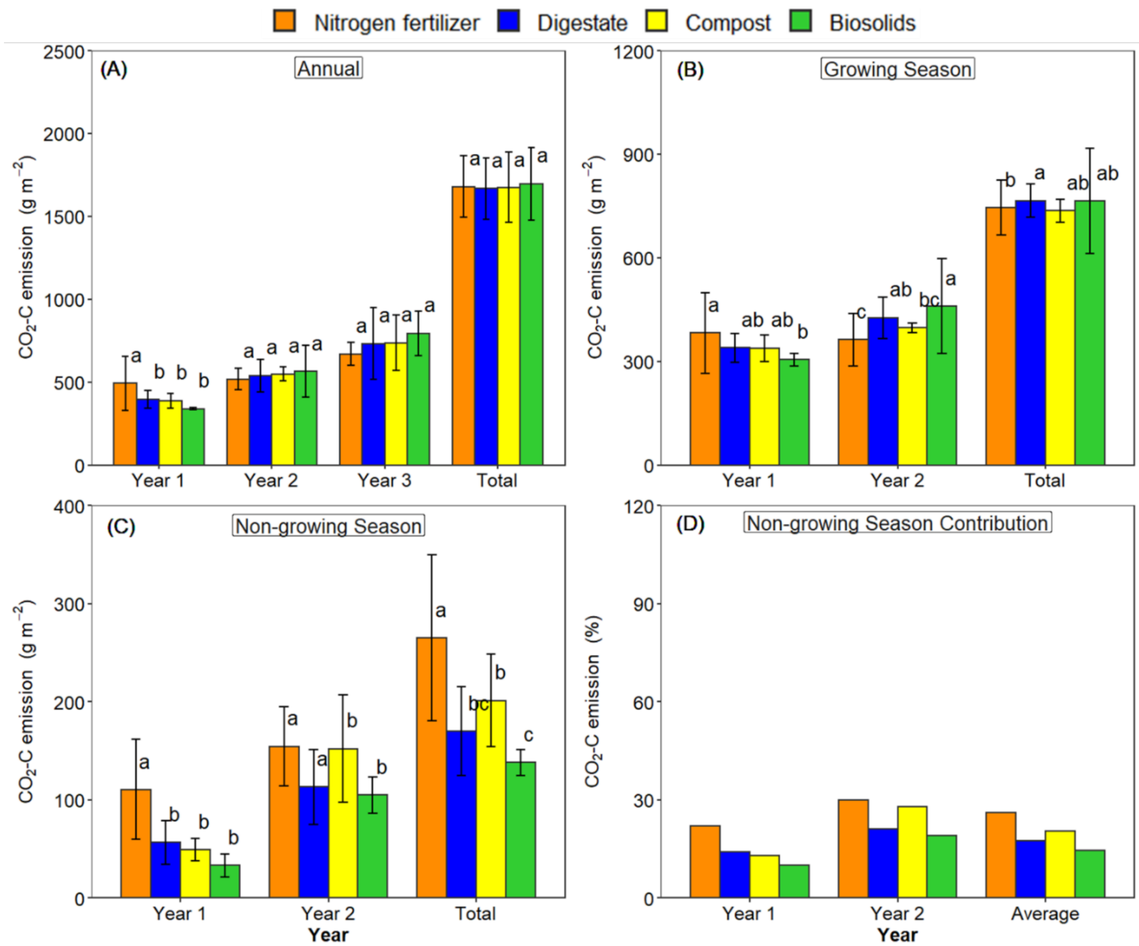


Figure 3.5: Cumulative carbon dioxide - CO_2 from soil amended with nitrogen fertilizer, digestate, compost, and biosolids in a corn-soybean rotation at Elora Research Station, Ontario, Canada. Letters indicated significant differences among the treatments (p<0.05).

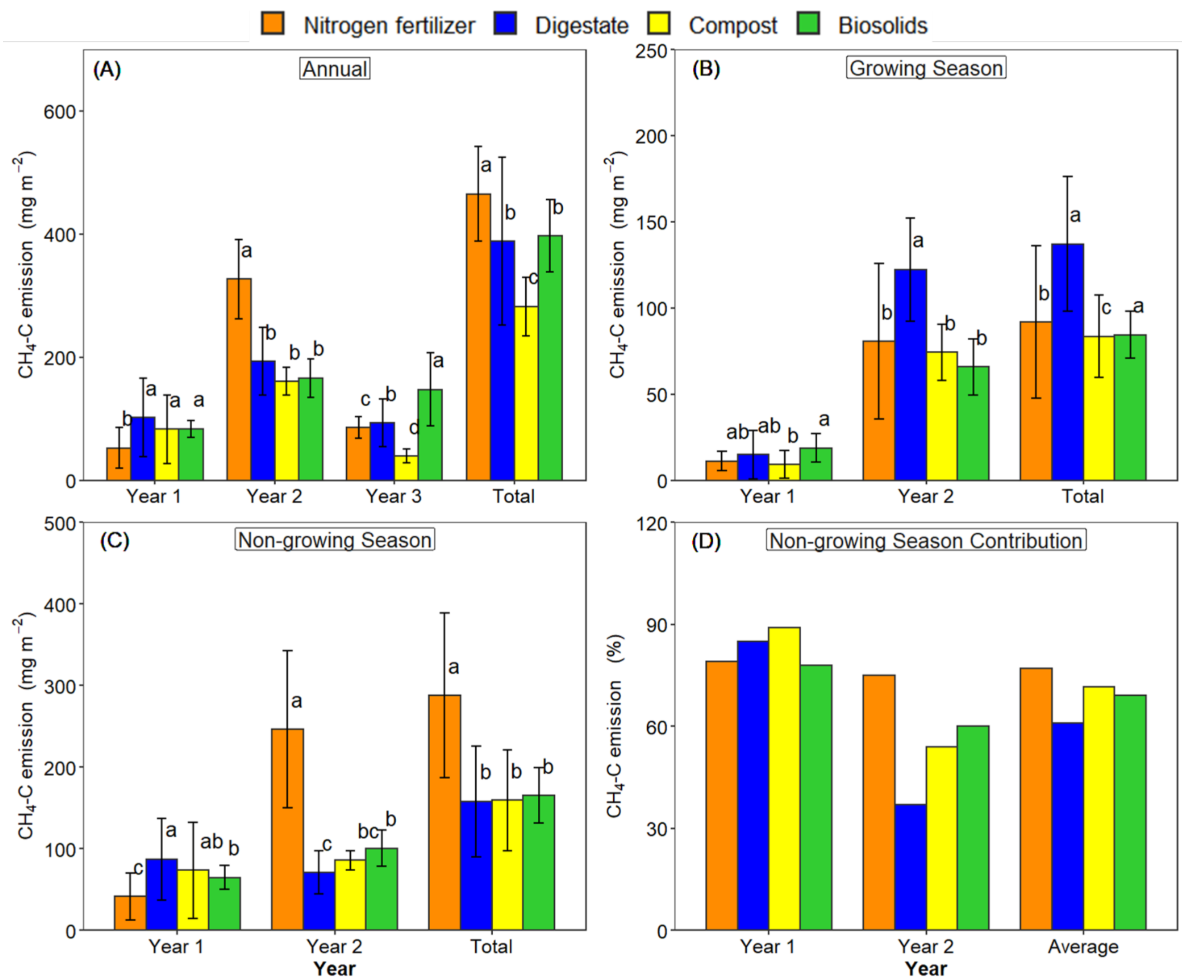


Figure 3.6: Cumulative methane - CH_4 emissions from soil amended with nitrogen fertilizer, digestate, compost, and biosolids in a corn-soybean rotation at Elora Research Station, Ontario, Canada. Letters indicated significant differences among the treatments (p<0.05).

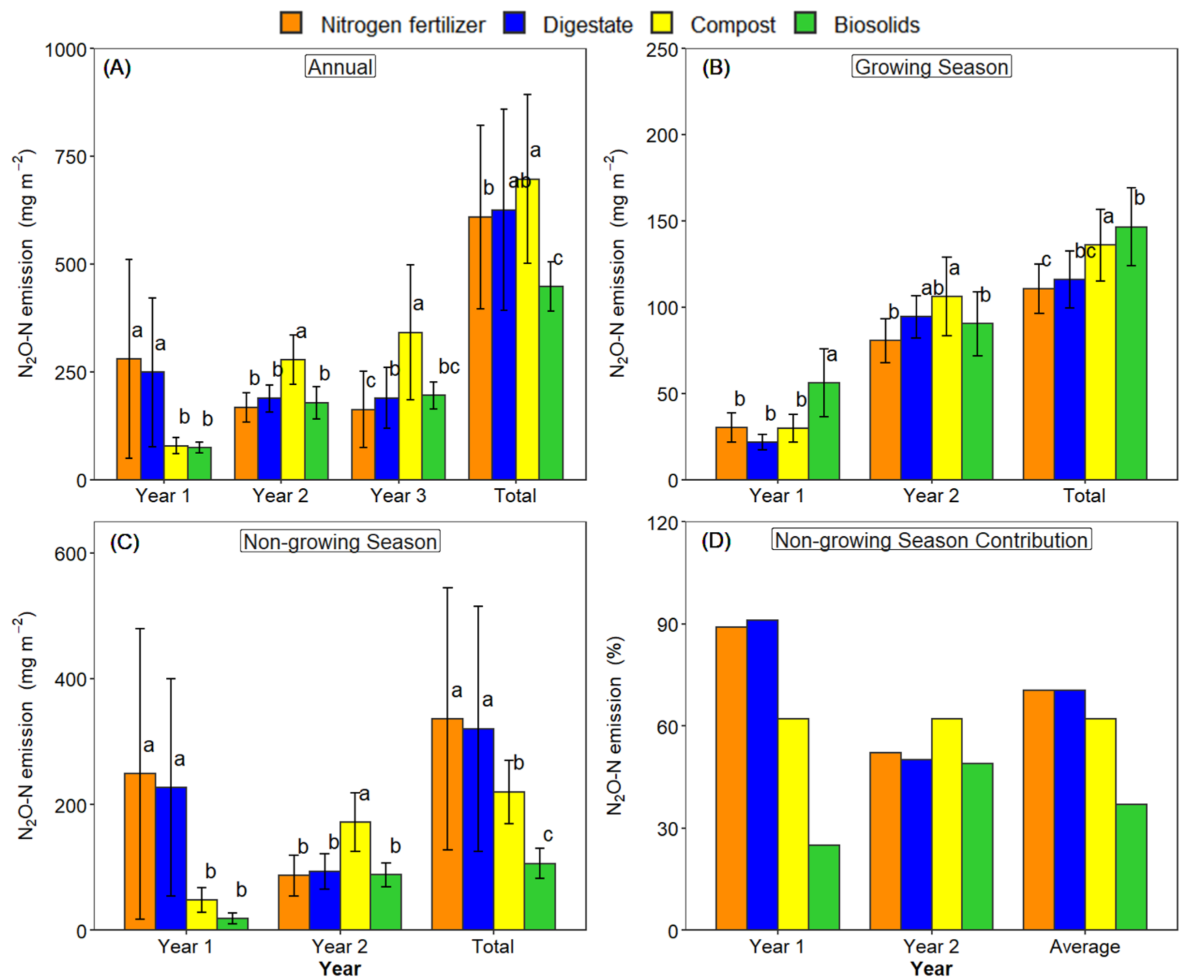


Figure 3.7: Cumulative nitrous oxide - N_2O emission from soil amended with nitrogen fertilizer, digestate, compost, and biosolids in a corn-soybean rotation at Elora Research Station, Ontario, Canada. Letters indicated significant differences among the treatments (p<0.05).

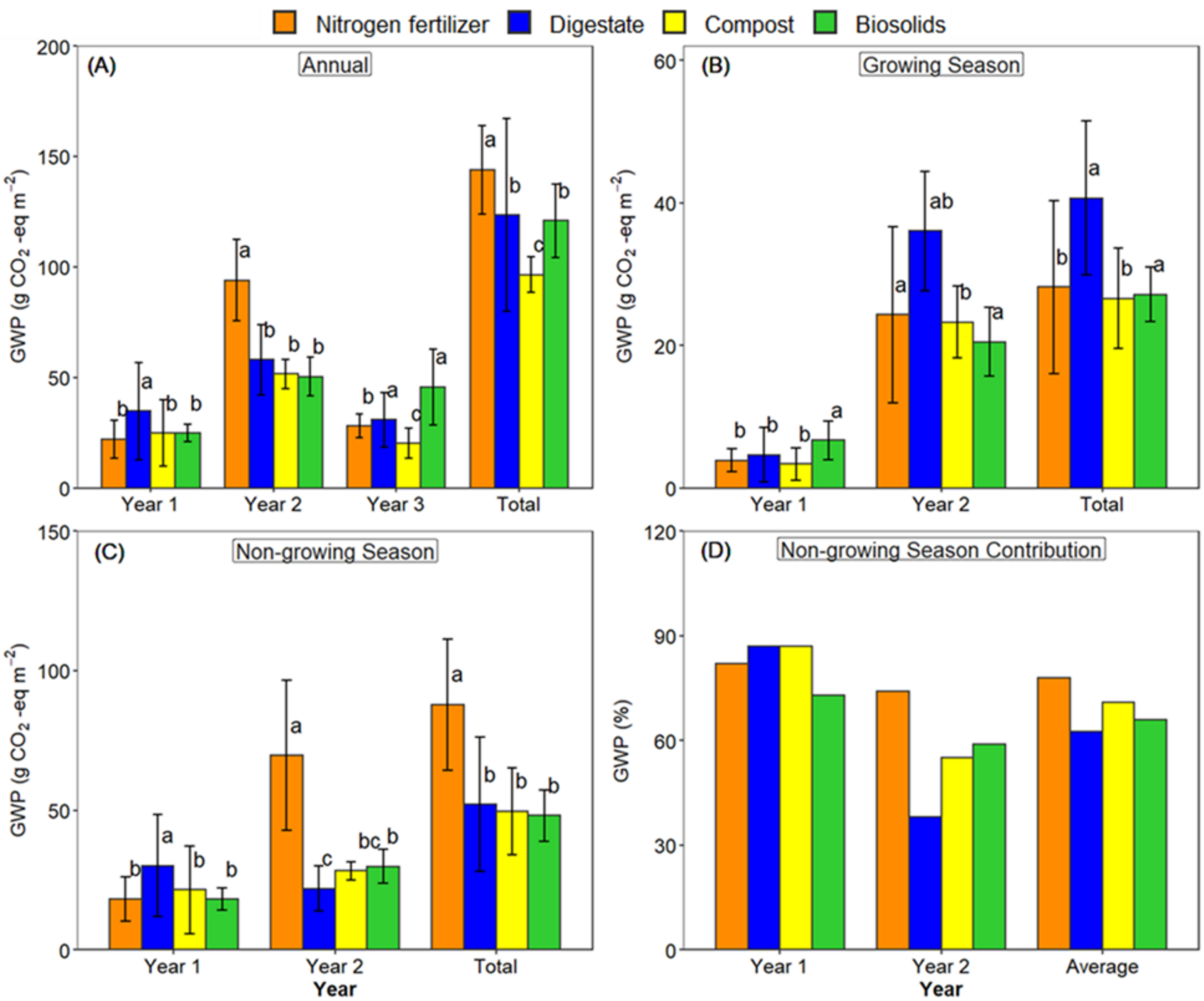


Figure 3.8: Global warming potential (GWP) from soil amended with nitrogen fertilizer, digestate, compost, and biosolids in a corn-soybean rotation at Elora Research Station, Ontario, Canada. Letters indicated significant differences among the treatments (p<0.05).

3.3.3 Emissions in relation to soil parameters

The general additive model determined that soil parameters that control the emissions during the growing and non-growing seasons were different (Table 3.3). Specifically, on an annual basis, soil moisture, temperature, and nitrate were significant predictors of soil CO_2 emission that accounted for 24% of the variability in annual CO_2 emissions (adj. R^2 = 0.22; Table 3.3). Only nitrate was the significant predictor for soil CH_4 emission, accounting for 5% of the variability in annual CH_4 emissions (adj. R^2 = 0.03; Table 3.3). Soil moisture, soil temperature, electrical conductivity, ammonium, and nitrate were significant predictors for soil N_2O emissions, accounting for 23% of the variability in annual N_2O emissions (adj. R^2 = 0.21; Table 3.3). During the non-growing season, no soil parameter was a significant predictor for emission except for soil temperature, soil moisture, and ammonium that drives N_2O emission.

The correlation between emissions and soil parameters varied with positive, negative, or no relationships (Table 3.4). The CO_2 emission had a weak positive correlation with soil temperature, nitrate, or electrical conductivity (p<0.05), a weak negative correlation with soil moisture or ammonium (p<0.05), and no significant correlation with CH_4 emission (p>0.05) (Table 3.4). The CH_4 emission had a weak positive correlation with soil moisture, a weak negative correlation with soil temperature or nitrate (p<0.05), and no significant correlation with electrical conductivity or ammonium (p>0.05) (Table 3.4). The N_2O emission had a moderate positive correlation with CO_2

emission, weak positive correlation with soil temperature, electrical conductivity, or nitrate (p<0.05), weak negative correlation with soil moisture or ammonium (p<0.05), and no significant correlation with CH_4 emission (p>0.05) (Table 3.4).

Table 3.3: Relationship between cumulative greenhouse gas emissions and soil parameters during corn-soybean rotation at Elora Research Station, Ontario, Canada. The functions s1 to s5 are the independent variables, Estimate (Est) is the effect of each of the variable, and edf is the effective degree of freedom of the model (edf =1 indicate a simple linear effect). p<0.05 are bolded.

Greenhouse gas Emission	Season	Soil parameters					Total model statistics
		s_1(SM)	s_2(ST)	s_3(EC)	s_4(NH_4)	s_5(NO_3)	
CO_2 Model	Annual	edf = 3.28 *p* = ***0.001***	edf =1.00 *p* = ***0.001***	edf =1.00 *p* = 0.061	edf =1.14 *p* = 0.158	edf =3.06 *p* = ***0.020***	*Est*= 113.59 *p* <***0.001***
		R^2 = 0.22, Deviance explained = 23.9%; n = 438					
	Growing Season	edf = 3.35 *p* = ***0.001***	edf =1.00 *p* = ***0.006***	edf =1.00 *p = 0.154*	edf =1.15 *p = 0.156*	edf =2.736 *p = 0.087*	*Est*= 127.94 *p* < ***0.001***
		R^2 = 0.18, Deviance explained = 19.7%; n = 390					
	Non-Growing Season	edf = 1.89 *p = 0.080*	edf =1.00 *p = 0.487*	edf =1.82 *p = 0.641*	edf =3.22 *p = 0.112*	edf =1.00 *p = 0.612*	*Est*= -3.06 *p = 0.538*
		R^2 = 0.36, Deviance explained = 47.7%; n = 48					
CH_4 Model	Annual	edf = 1.28 *p = 0.202*	edf =1.00 *p = 0.916*	edf =1.01 *p = 0.202*	edf =2.48 *p = 0.200*	edf =1.00 *p* = ***0.022***	*Est*= -35.33 *p* = ***0.001***
		R^2 = 0.03, Deviance explained = 4.63%; n = 436					
	Growing Season	edf = 1.30 *p=0.190*	edf =1.00 *p = 0.575*	edf =1.00 *p = 0.197*	edf =2.55 *p = 0.127*	edf =1.00 *p* = ***0.026***	*Est*=-35.33 *p* = ***0.001***
		R^2 = 0.03, Deviance explained = 4.72%; n = 388					
	Non-Growing Season	edf = 1.60 *p = 0.764*	edf =1.00 *p = 0.426*	edf =1.00 *p = 0.610*	edf =1.00 *p = 0.091*	edf =1.44 *p = 0.293*	*Est*= -2.99 *p = 0.395*
		R^2 = 0.16, Deviance explained = 26.7%; n = 48					
N_2O Model	Annual	edf = 4.06 *p* < ***0.001***	edf =1.00 *p* = ***0.003***	edf =1.00 *p* = ***0.043***	edf =1.50 *p* = ***0.003***	edf =1.00 *p* = ***0.008***	*Est* =16.45 *p* = ***0.001***
		R^2 = 0.21, Deviance explained = 22.9%; n = 436					
	Growing Season	edf =3.97 *p* = ***0.001***	edf =1.00 *p* = ***0.006***	edf =1.01 *p* = 0.055	edf =1.68 *p* = ***0.015***	edf =1.00 *p* = ***0.016***	*Est* = 20.69 *p* = ***0.001***
		R^2 = 0.18, Deviance explained = 20.2%; n = 389					
	Non-Growing Season	edf = 3.29 *p* = ***0.002***	edf =1.00 *p* = ***0.028***	edf =1.83 *p* = 0.235	edf =4.78 *p* = ***0.001***	edf =1.00 *p* = 0.716	*Est*= -18.66 *p* = ***0.012***
		R^2 = 0.55, Deviance explained = 66.8%; n = 47					

SM refers to soil moisture, ST refers to soil temperature, and EC refers to electrical conductivity, NH_4 refers to ammonium, NO_3 refers to nitrate.

Table 3.4: Correlation values for the greenhouse gas fluxes and emissions and soil parameter during the growing and non-growing season of 2018 to 2020 at Elora Research Station, Ontario, Canada (n=432).

Variables	N_2O (μg m^{-2})	CO_2 (mg m^{-2})	CH_4 (μg m^{-2})	SM (%)	ST (^oC)	EC (mS m^{-1})	NH_4^+-N (mg kg^{-1})	NO_3^--N (mg kg^{-1})
N_2O	1							
CO_2	0.49**	1						
CH_4	0.03	-0.06	1					
SMC	-0.17**	-0.32**	0.12*	1				
ST	0.26**	0.39**	-0.11*	-0.54**	1			
EC	0.21**	0.1*	-0.03	-0.33**	0.43**	1		
NH_4^+-N	-0.13**	-0.09*	0.06	-0.05	-0.07	0.19**	1	
NO_3^--N	0.3**	0.26**	-0.14**	-0.43**	0.55**	0.62**	0.06	1

'*' Significant at the .05 probability level. '**' Significant at the .01 probability level. Note: SM refers to soil moisture, ST refers to soil temperature, and EC refers to electrical conductivity.

3.4 Discussion

3.4. 1 Greenhouse gas fluxes and cumulative emissions

Although the greenhouse gas fluxes were not different among the treatments, cumulative emissions varied remarkably. Similarly, Adjuik et al. (2020) found greenhouse gas fluxes were not significantly different among fertilization treatments that used digestate and nitrogen fertilizer (urea). The cumulative emissions varied among the treatments except for CO_2 emission. Similarly, Obi-Njoku et al. (2022) found that N_2O emission was significantly different among treatments - anaerobically digested biosolids, composted biosolids, alkaline-stabilized biosolids, and urea. They also found that CO_2 emission was similar among the treatments. Roman-Perez et al. (2021) also found CH_4 emission was different among treatments - mesophilic anaerobic digested biosolids, alkaline-stabilized biosolids, composted biosolids, and urea. Cumulative N_2O emission variation among treatments was probably due to the carbon and nitrogen content of biobased residues influenced by the source and production method. For the experiment, the biosolids used was a thermal hydrolyzed semi-solid with disintegrated complex carbon compounds that likely contain fewer denitrifying bacteria for greenhouse gas production (Lystek, 2021, Murray et al., 2019). The digestate used was a liquid form of organic material with a low soluble organic carbon content with a probably low energy source for denitrifiers responsible for N_2O emission (Vallejo et al., 2006; Nkoa, 2014). In addition, cumulative CH_4 emission varied among the treatments, and nitrogen fertilizer had the highest CH_4

emission. This was likely due to the lower rate of nitrogen applied in the nitrogen fertilizer treatment compared to that of biobased residue treatments (Table 3.2). This result corroborates that CH_4 emissions increase with lower fertilizer nitrogen rates. The review by Linquist et al. (2012) established that CH_4 emission increase with low nitrogen rates of either urea or ammonium sulfate, while high nitrogen rates – applied beyond crop demand decrease CH_4 emission. Yao et al. (2012) also found that nitrogen fertilizers, particularly urea-based fertilizers, inhibit CH_4 emission in a sandy loam rice paddy field.

The influence of soil temperature, electrical conductivity, nitrate, ammonium and soil moisture varied among the greenhouse gas emissions. Soil temperature, electrical conductivity and nitrate were found to have a positive relationship with CO_2 emission while soil moisture have a negative relationship with CO_2 emission (Table 3.3, 3.4). This was likely due to the availability of nutrients, substrate, and warming condition that enhanced microbial activity causing CO_2 production (Soosaar et al., 2011; Mori et al., 2017). Also, similar with other studies (Lu et al., 2014; He et al., 2016, Tang et al., 2021), the negative relationship between soil moisture and CO_2 was due to the high moisture during sampling time and the transient waterlogged conditions. This is because waterlogged or saturated conditions increase diffusional resistance that reduces soil CO_2 emission (Ben-Noah and Friedman, 2018). CH_4 emission had a significant positive relationship with soil moisture, negative relationship with soil temperature and nitrate (Table 3.3, 3.6). This implied there was sufficient moisture level that enhanced the abundance of methanogens through methanogenesis for CH_4 production (Kuzyakov and Xu, 2013; Song et al., 2021). As soil temperature decreased CH_4 emissions increased due to soil drying (Maljanen et al., 2003; Zhao et al., 2019), therefore confirming that a negative relationship between CH_4 and soil temperature or that of soil temperature with soil moisture (Table 3.7). Furthermore, nitrate was a key regulator of CH_4 emissions (Table 3.3) probably due to the varied nitrate supply among the biobased residues that favour denitrifying bacteria instead of methanogens that are responsible for CH_4 production (Le Mer and Roger, 2001; Thangarajan et al., 2013).

Soil temperature, electrical conductivity, nitrate have a positive relationship with N_2O emission indicating that soil temperature stimulated nitrogen mineralization and increase inorganic nitrogen availability, providing substrate for nitrification and denitrification (Cui et al., 2018; Li et al., 2020). Addition of nitrogen fertilizer increased soil electrical conductivity which supports microbial nitrogen transformations towards N_2O production pathways (Adviento-Borbe et al., 2006). The increase of nitrate and decrease of ammonium as N_2O emission increase suggests that denitrification may play a more important role in soil N_2O emissions than nitrification (Zheng et al., 2021) because soil nitrate as a substrate source for denitrification induces more N_2O emission (Table 3.3, 3.6). However, soil nitrate and electrical conductivity were not key drivers during the non-growing season because they were lower compared to growing season. This chapter also showed strong positive correlation of nitrate and electrical conductivity (Table 3.4).

Equally, Baral et al. (2022) found that due to low nitrate concentration (<12 mg kg^{-1}) soil nitrate was not a determining factor during non-growing season. Related studies such as (Adviento-Borbe et al., 2006; Lu et al., 2018) also established a linear relationship of nitrate and electrical conductivity corresponds to N_2O emission.

Furthermore, accounting for non-growing season CO_2 emission is necessary for yearly budget and uncertainty assessment between agricultural soils and climate systems. The contribution of non-growing season CO_2 to its annual emission (range; 13–30%) was higher compared to other studies. Miao et al. (2014) found that non-growing season CO_2 emission accounted for about 6 – 7% of annual emission among fertilization treatments that used nitrogen fertilizer, plus composted pig manure on frozen cropland in northeast China. The contribution of non-growing season CH_4 to its annual emission was between 37% to 89%, suggesting its importance to contributing to greenhouse gas fluxes from temperate agricultural landscapes. This shows that high CH_4 emissions occur during the non-growing season in the three years crop-soybean rotation under different nitrogen fertilizer management in an agricultural soil. Thus, the addition of fertilizer and amendment does not necessarily support previous research that suggested that agricultural soils consume CH_4 due to the used nitrogen fertilizer in addition to the emission of CH_4 from the amendment itself during the non-growing season (Basta et al., 2001; Smith et al., 2011).

The contribution of non-growing season N_2O to its annual emission (range; 25 – 91%) was similar to the value reported in other studies. For example, Chantigny et al. (2016) found that non-growing season N_2O emission accounted for about 16 - 79% of annual emission for silty clay and 46 - 87% for the sandy loam soil. Yanai et al. (2011) found that 58 - 85% of the annual emission accounted for non-growing season N_2O emission from arable lands in northern Japan. The relatively varied and high contribution of non-growing season N_2O was likely due to the freeze thaw events that could stimulate N_2O pulse through microbial metabolism and solid-liquid gas diffusivity (Kim et al., 2012; Congreves et al., 2018; King et al., 2021). Also, the legacy carbon and nitrogen derived from crop residues and/or from nitrogen fertilizer application (Figure 3.4) attested to the availability of substrates for the microbial community to produce N_2O. In addition, biobased residues decreased N_2O emission mainly in the non-growing season but not in the growing season (Figures 3.7) due to carbon and nitrogen input from the biobased residues (Table 3.2) that supports microbial metabolism and soil nitrogen accumulation more in the growing season than non-growing season (Figure 3.4). The low nitrogen accumulation during the non-growing season for biobased residues when compared to nitrogen fertilizer (Figure 3.4) further illustrated that a low substrate availability slows the microbial processes that produces N_2O (Rousset et al., 2022).

3.4.2 Global warming potential (GWP)

Global warming potential for nitrogen fertilizer was substantial compared to that of biobased residues indicating nitrogen fertilizer has the highest net source of N_2O and CH_4 emissions. Shen et al. (2018) found that nitrogen fertilizer inputs were responsible for 26% to 74% of the indirect global warming potential. Consistent with the finding of Dendooven et al. (2012), N_2O emission contributed most to the global warming potential. The global warming potential ranged from 22 to 94 g CO_2-eq m^{-2} yr^{-1} under the maize-soybean rotation using the Intergovernmental Panel of Climate Change default emission coefficients and only N_2O and CH_4 emissions. Similarly, Adviento-Borbe et al. (2007) adopted the approach and found global warming potential ranging from 54 to 102 g CO_2-eq m^{-2} yr^{-1}. However, this chapter's results were limited to net source with no net sink determination for CH_4 and N_2O due to the zero assumption for negative emission during cumulative emission calculation. Therefore, further research should consider evaluating the net sink potential of the biobased residues.

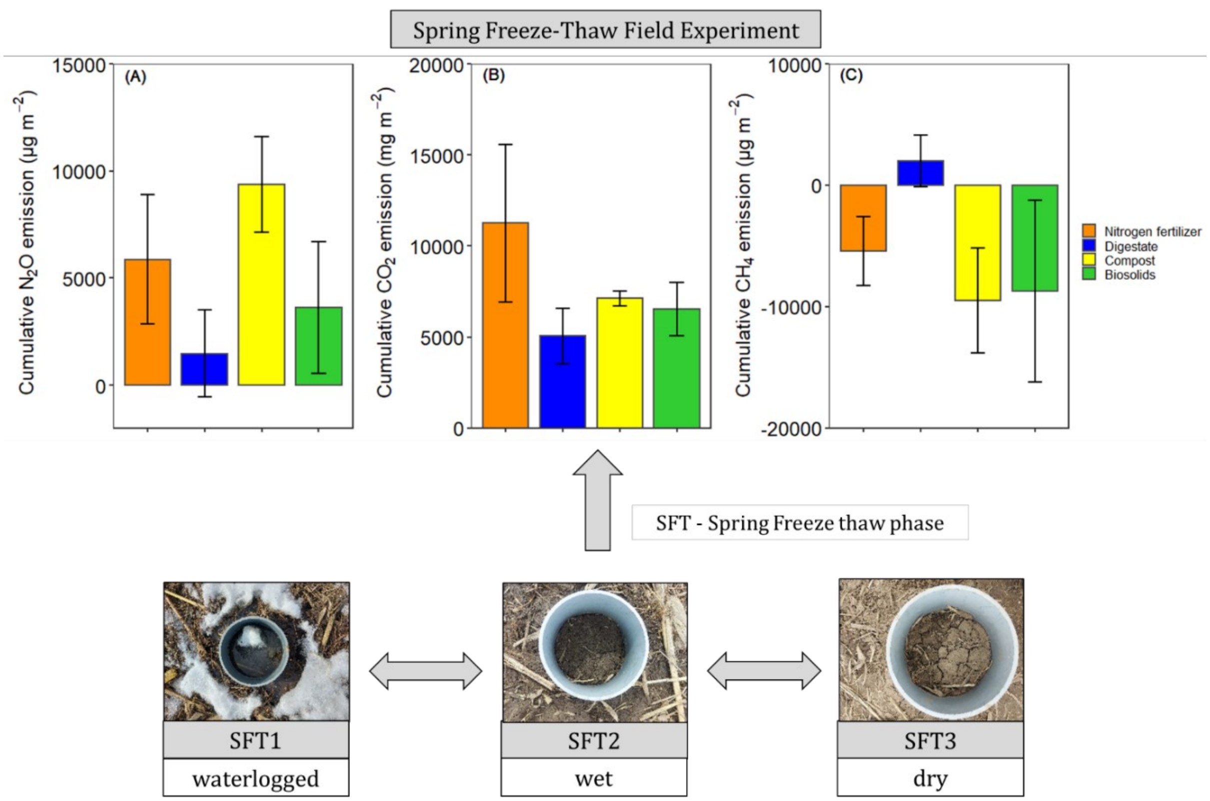
3.5 Conclusions

The impact of biobased residues on greenhouse gas emissions during the growing and non-growing seasons determined that N_2O emission from compost was highest, while biosolids was the lowest compared to the nitrogen fertilizer. Biobased residues have lower CH_4, and global warming potential compared to nitrogen fertilizer, while no difference between the CO_2 emission among the treatments. Non-growing season emissions contributed significantly to the annual budget. Soil parameters measured in this chapter sufficiently account for the emission controlling factors for N_2O and CO_2 but not CH_4. Further studies should incorporate automatic continuous flux monitoring methods such as opaque chambers equipped with near-infrared laser CH_4 analyzer and middle infrared laser N_2O analyzer. This will help understand the capacity of biobased residues explicitly to serve as net sink balance, especially during the non-growing season, and the relative contribution of the greenhouse gases at a specific endpoint in time.

Chapter 4

Spring freeze-thaw stimulates greenhouse gas emissions from agricultural soil

Graphical Abstract



4.1 Introduction

In temperate cold regions, spring freeze-thaw events contribute to the production of greenhouse gases, which further exacerbates climate change (Pelster et al., 2013; Natali et al., 2019). Greenhouse gas production during spring freeze-thaw account for 29-73% of the annual N_2O (Hung et al., 2021), 5-50% of the annual CO_2 (Kurganova et al., 2007), and ~80% of the annual CH_4 (Song et al., 2012). In temperate agroecosystems, greenhouse gas fluxes peak during spring freeze-thaw events due to carbon and reactive nitrogen remaining in the soil from the previous crop season (i.e., legacy carbon and nitrogen) (Wagner-Riddle et al., 2017). Accordingly, amending the soil with organic residues, high in carbon and nitrogen, increases the production and fluxes of greenhouse gases considerably during spring freeze-thaw events (Kariyapperuma et al., 2012; Chantigny et al., 2016). Furthermore, intermittent greenhouse gas fluxes during freeze-thaw events are also related to temporal changes in soil biophysical conditions influenced by snow melting, waterlogging, and gradual warming and drying (Congreves et al., 2018).

Biobased residues are biological wastes that include but are not limited to, compost (e.g., food waste), biosolids, and anaerobic digestate. Biobased residues are comprised of readily accessible organic carbon and nitrogen, thereby enhancing soil microbial activity and greenhouse gas productions (Thangarajan et al., 2013). Furthermore, organic matter, nutrient concentration, as well as physical characteristics (e.g., composts are solid, biosolids are semi-liquid and anaerobic digestates are liquid) can vary substantially among biobased residue types (Bertora et al., 2008; Grave et al., 2018, Badewa and Oelbermann, 2020). Differences in biobased residue composition also influence the soil carbon and nitrogen dynamics and microbial activity, which regulates greenhouse gas production (Miura et al., 2019; Rosinger et al., 2022).

Since spring freeze-thaw influences the soil's biophysical conditions, the presence of readily available carbon and nitrogen substrates to the microbial community from biobased residues can cause nitrification, incomplete denitrification or methanogenesis, resulting in the generation of greenhouse gases (Zhu-Barker et al., 2015; Brenzinger et al., 2018). Additionally, temperature fluctuation and moisture variation (waterlogged, wet, and dry) during spring freeze-thaw, where the soil has minimal or no snow cover, causes frequent and intense episodes of freezing and thawing coupled with physical deformation of soil aggregates (Miao et al., 2014; Wu et al., 2017; 2021). Furthermore, substantial soil warming during the thaw phase increases the microbial activity and triggers greenhouse gas emissions (Henry, 2008, 2013; Chantigny et al., 2019).

Although multiple studies determined the impact of biobased residues on greenhouse gas productions (Thangarajan et al., 2013; Inslam et al., 2015; Roman-Perez et al., 2021), the knowledge on the impact of biobased residues on spring freeze-thaw events and resultant greenhouse gas fluxes remains limited. However, knowing the contribution of biobased residues to greenhouse gas fluxes is critical since biological and physicochemical properties, in addition to soil management (e.g., the rate and timing of biobased residue application, soil cultivation) and environmental characteristics (e.g., soil type, weather), vary at the local and regional scale (Dambreville et al., 2008; Eckard et al., 2010; Grave et al., 2018). Although some studies found that freeze-thaw events can cause a shift in soil physical conditions and how biobased residues influence this shift and respond to greenhouse gas production, but they have only been evaluated under laboratory conditions and with inconsistent results (Sahin et al., 2008; Liu et al., 2016; Fu et al., 2019; Pupko, 2019, Hou et al., 2020). Under field conditions, Roman-Perez et al. (2021) found that soil amended with contrasting biosolids types (composted, anaerobic digested, alkaline stabilized) varied in their production of N_2O. Roman-Perez et al. (2021) also found that N_2O fluxes from biobased residues were higher compared to those from nitrogen fertilizer. Consequently, knowledge on the impact of biobased residues on greenhouse gas production during spring freeze-thaw events in a field setting is warranted (Hung et al., 2021).

The objectives of this chapter were to quantify and compare greenhouse gas fluxes from soil amended with biobased residues and nitrogen fertilizer during the spring freeze-thaw events. This chapter hypothesizes that (i) N_2O, CO_2 and CH_4 fluxes from soil amended with biobased residues will be greater compared to soil amended with nitrogen fertilizer during spring freeze-thaw since a greater quantity of carbon and nitrogen is available from the biobased residues than from nitrogen fertilizer. (ii) N_2O and CH_4 fluxes will be greater during the waterlogged and wet phase due to denitrification and methanogenesis caused by anaerobioses, whereas the CO_2 flux will be greater during the dry phase due to enhanced microbial activity caused by a rising soil temperature (iii) CO_2 flux is strongly related to soil temperature throughout the spring freeze-thaw.

4.2 Materials and Methods

4.2.1 Site description and field experiment

See Chapter 3, section *3.2.1 Experimental site, soil, and biobased residues* and section *3.2.2 Experimental Design and management.* The chapter, however, focused on the impact of spring freeze-thaw event on the greenhouse gas emission of the four treatments applied in May 2020 during the spring of 2021 (March specifically). At this field location, soils undergo spring freeze-thaw from March to April, when snow begins to melt. This creates an environment with transitory waterlogging, a partial ice layer, and wet and drying conditions (Table S4.1). Daily ambient temperature fluctuations during the spring cause repeated freeze-thaw cycles, and the daily average maximum and minimum temperatures in this site were 6.9 to −3.2 °C between March 1 and April 30 from 2004 to 2020 (Environment and Climate Change Canada, 2021).

The effect of ambient temperature, snow depth, and precipitation were obtained from a weather station located at the research station. Soil temperature at ∼10 cm depth was measured by temperature sensors buried in one replicate of each treatment (n = 4).

Air and soil temperature followed a similar pattern from October to mid-November but were distinctly different from the end of November to mid-March when the soil was covered with snow for 95 days (Figure S4.1). During this period, the snow cover (~49 cm) and crop residues left on the soil insulated and regulated the soil temperature (0-10 cm depth) to ~0^oC with precipitation ranging from ~29 mm (Figure S4.1)

4.2.2 Greenhouse gas fluxes

See Chapter 3, section *3.2.3 Greenhouse gas fluxes*

Greenhouse gas fluxes were measured from 11 March to 24 March 2021 using closed static chambers. The first measurement occurred on March 11, 2021, when the chamber bottom in the treatment plots were visible - not covered with snow and had <5 cm of snow. The final measurement occurred on March 24, 2021, 7 days after the snow and ice melted disappeared. Due to freezing soil state and data logger issues, moisture was not recorded. Hence, the measurement days was categorized into three freeze-thaw phases using the visual moisture state observed during greenhouse gas measurement (Table S4.1). Qualitative moisture assessment occurred within 1 m around the chamber for greenhouse gas measurement since soil is heterogenous with different visual moisture state. First freeze-thaw phase (waterlogged) occurred when the soil was covered with melting snow and saturated. The second freeze-thaw phase (wet) occurred immediately after the snow and ice melted and when mostly wet, followed by third freeze-thaw phase (dry) that was mostly dry with wet patches (Table S4.1).

Gas samples were extracted daily from each treatment replicate. Cumulative emission of the gases and the global warming potential over 100 years (GWP) during the spring freeze-thaw events (11 March to 24 March 2021) were calculated using equations (4.1) and (4.2).

$$CE = \sum F_i \text{ x } 24 \tag{4.1}$$

where CE is the cumulative emission of soil greenhouse gas (μg m^{-2}/ mg m^{-2}) and Fi is the greenhouse gas flux of the i-th sampling time (μg m^{-2} h^{-1}). The GWP (kg CO_2 eq ha^{-1}) was calculated using:

$$GWP = CE_{CO_2} + CE_{CH_4} \text{ x } 27.9 + CE_{N_2O} \text{ x } 273 \tag{4.2}$$

where CE_{CO_2}, CE_{CH_4}, and CE_{N_2O} represent the cumulative emissions of CO_2, CH_4, and N_2O, respectively.

4.2.3 Statistical analysis

All data processing and analysis were carried out in R v.4.2.1 (R Core Team, 2020). Data did not meet assumption of normality, so data were analyzed by non-parametric statistics. The effect of treatment and the freeze-thaw phases on greenhouse gas flux and emission was determined with a Kruskal Wallis test and the treatment comparison was done with a Dunn test (p<0.05). The relationship between greenhouse gas emission and soil or air temperature was determined using Pearson correlation coefficients (r). The relationship between greenhouse gas fluxes and soil temperature was evaluated using Pearson correlation coefficients (r) and exponential model.

4.3 Results

4.3.1 Temperature conditions and greenhouse gas fluxes

Soil temperature among treatments was identical ($p>0.05$) during the spring freeze-thaw (Figure 4.1). Soil and air temperature varied among the freeze-thaw phases ($p<0.05$) with average soil temperature during greenhouse gas sampling highest for dry (7^oC), followed by wet (2^oC) and lowest for waterlogged (0^oC) (Figure 4.1). Cumulative N_2O emission were different among treatments ($p=0.020$), while cumulative CO_2 and CH_4 emissions were not different among treatments ($p>0.05$) (Table 4.1). When comparing N_2O emission among treatments, the average loss was highest in compost (669 µg N_2O-N m^{-2}), followed by nitrogen fertilizer (419 µg N_2O-N m^{-2}), biosolids (259 µg N_2O-N m^{-2}), and digestate (105 µg N_2O-N m^{-2}) ($p<0.05$). The global warming potential varied among treatments but was not significantly different ($p>0.05$), where nitrogen fertilizer had the highest (127 kg CO_2 ha^{-1}) and digestate the lowest (55 kg CO_2 ha^{-1}) warming potential (Table 4.1). The greenhouse gas emission varied significantly among the freeze thaw phases ($p<0.05$) with peak N_2O and CO_2 fluxes during the wet, and peak CH_4 flux during the dry (Figure 4.2). N_2O flux ranged from -388 to 454 µg N_2O-N m^{-2} h^{-1}, CO_2 flux from -211 to 565 mg CO_2 -C m^{-2} h^{-1}, and CH_4 flux -1285 to 216 µg CH_4-C m^{-2} h^{-1}. However, mean N_2O flux was highest during the wet (31 µg N_2O-N m^{-2} h^{-1}) and lowest during waterlogged (-2 µg N_2O-N m^{-2} h^{-1}), mean CO_2 flux was highest flux during the dry (34 mg CO_2 -C m^{-2} h^{-1}) and lowest during waterlogged (5 mg CO_2 -C m^{-2} h^{-1}) while mean CH4 flux was highest during the wet (-11 μg CH_4-C m^{-2} h^{-1}) and lowest during waterlogged (-24 μg CH_4-C m^{-2} h^{-1}).

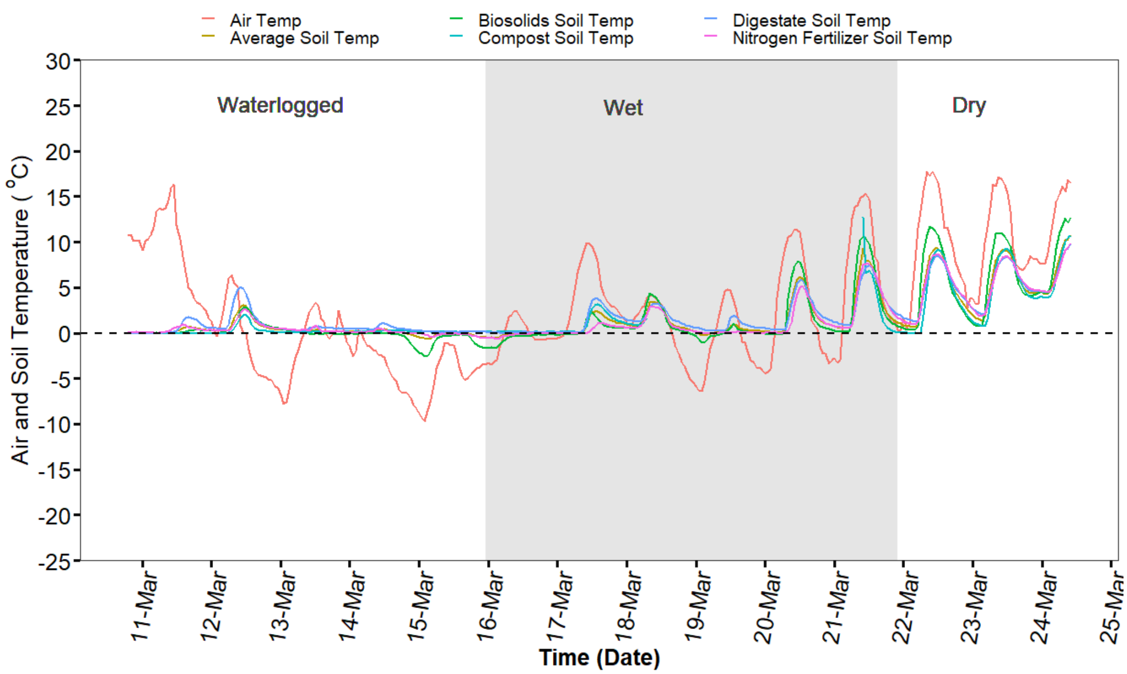


Figure 4.1: Hourly soil temperature from the four treatment plots (~10cm) and air temperature during the experiment period (from 11 March 2020 to 24 March 2021) at Elora Research Station, Ontario, Canada.

Table 4.1: Cumulative greenhouse gas emission and global warming potential from soils amended with nitrogen fertilizer and biobased residues during the 2021 spring freeze-thaw event at Elora Research Station, Ontario, Canada.

Treatment	**N_2O cumulative emission ($\mu g\ m^{-2}$)**	**CO_2 cumulative emission ($mg\ m^{-2}$)**	**CH_4 cumulative emission ($\mu g\ m^{-2}$)**	**GWP ($kg\ CO_{2\ eq}\ ha^{-1}$)**
Nitrogen fertilizer	5859 ± 3017^{b}	11255 ± 4318^{a}	-5415 ± 2824^{a}	127 ± 50^{a}
Compost	9372 ± 2237^{a}	7124 ± 416^{a}	-9487 ± 4320^{a}	94 ± 9^{a}
Biosolids	3626 ± 3086^{b}	6532 ± 1482^{a}	-8714 ± 7476^{a}	73 ± 19^{a}
Digestate	1468 ± 2028^{b}	5052 ± 1520^{a}	1971 ± 2136^{a}	55 ± 14^{a}

Mean with the same letter are not significantly difference (p>0.05) among treatments.

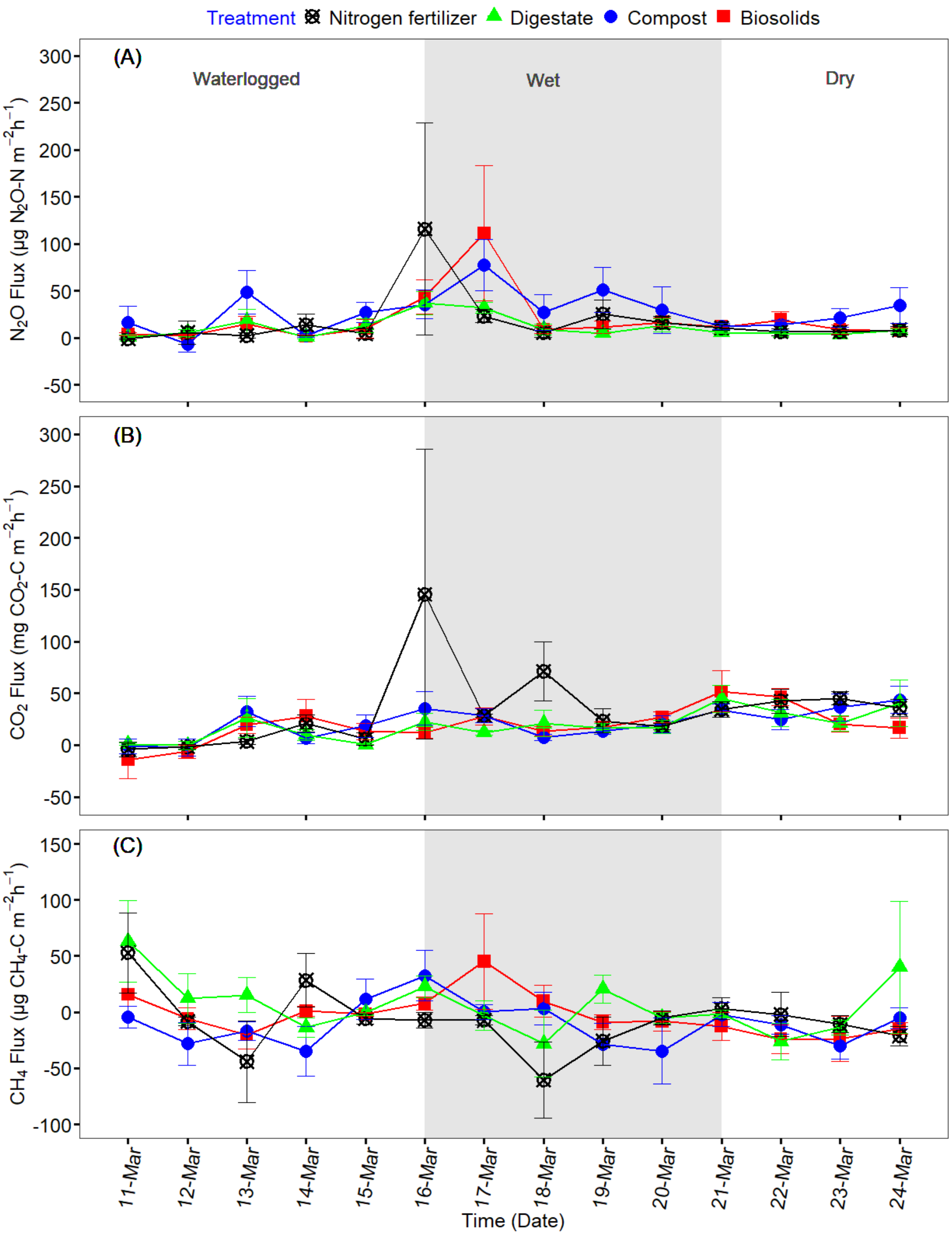


Figure 4.2: Greenhouse gas fluxes of N_2O, CO_2, and CH_4 during the spring freeze-thaw event of 11 March to 24 March 2021 at Elora Research Station, Ontario, Canada.

4.3.2 Relationship between greenhouse gas emissions, air, and soil temperature

N_2O emission had a moderate positive correlation with CO_2 emission (p=0.000), a weak positive correlation with CH_4 emission (p=0.006), but a weak non-significant correlation with soil or air temperature (p>0.05) (Table 4.2). CO_2 emission had a weak positive correlation with soil temperature (p=0.032) and a weak positive correlation with air temperature (p=0.042) but was not significantly correlated with CH_4 emission (p>0.05) (Table 4.2). CH_4 emission was not significantly correlated to CO_2 emission and soil and air temperature (p>0.05) except CH_4 emission (p=0.006) (Table 4.2). The relationship between greenhouse gas fluxes and soil temperature was estimated using an exponential model (Figure 4.3, Table 4.3). The CO_2 flux was positively correlated with soil temperature for compost, biosolids, and digestate when averaging all four treatments (p<0.05) (Figure 4.3a). However, N_2O and CH_4 fluxes were generally not significantly correlated with soil temperature (p>0.05) (Figure 4.3b, c).

Table 4.2: Correlation values for the greenhouse gas fluxes and emissions and soil and ambient temperature during the spring freeze-thaw events at Elora Research Station, Ontario, Canada. (n=224).

Variable	N_2O (µg m^{-2})	CO_2 (mg m^{-2})	CH_4 (µg m^{-2})	Soil Temperature (°C)	Air Temperature (°C)
N_2O	1				
CO_2	0.65**	1			
CH_4	0.18**	0.01	1		
Soil Temperature	-0.12	0.14*	-0.13	1	
Air Temperature	-0.08	0.08	0.02	0.74**	1

'*' Significant at the .05 probability level. '**' Significant at the .01 probability level.

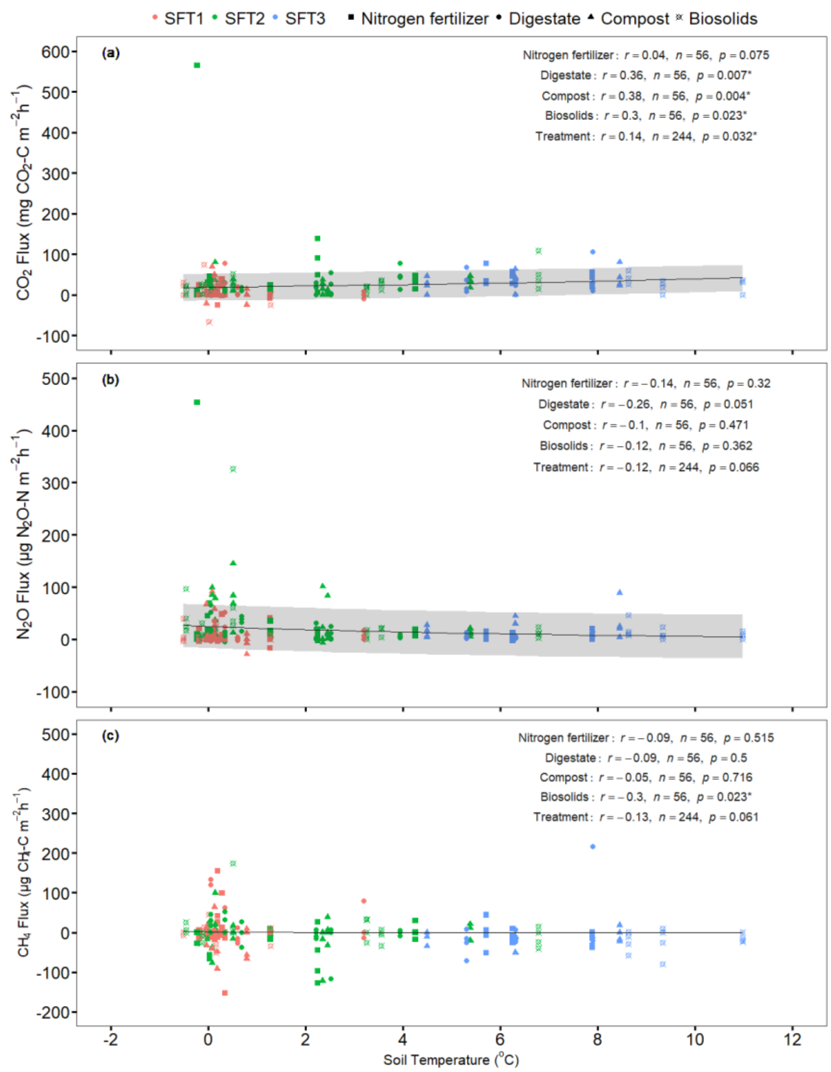


Figure 4.3: The effect of soil temperature (~10 cm) on (a) CO_2 (b) N_2O (c) CH_4 fluxes released from the soil amended with nitrogen fertilizer and biobased residues during the spring freeze-thaw event from 11 March to 24 March 2021. The graph presented the exponential model fitted (solid gray line) and the standard deviation of an exponential model fitted (gray shading) between greenhouse gas flux per treatment and soil temperature during the spring freeze-thaw event (note: CH_4 flux standard deviation was not achieved due to iterative error caused by the many negative integers). The Pearson correlation coefficients (r) and the significant level (p) are shown for the treatments. SFT1 is the first spring freeze-thaw phase (waterlogged), SFT2 is the second spring freeze-thaw phase (wet), and SFT3 is the third spring freeze-thaw phase (dry). * Shows p-value < 0.05.

Table 4.3: Relationship between greenhouse gas fluxes and soil temperature (~10 cm) using exponential model during the 2021 spring freeze-thaw events at Elora Research Station, Ontario, Canada.

Flux vs. Soil Temperature	**Equation: Soil_flux = *a* * exp (b * Soil_temp)**				
	Coefficients	Estimate	Std. Error	*t* value	Pr(>|*t*|)
Soil CO_2_flux	*a*	19.030	3.480	5.468	<0.001
	b	0.072	0.034	2.138	0.034
		Residual standard error= 43.39;			*n* = 222
Soil N_2O_flux	*a*	24.626	3.957	6.223	<0.001
	b	-0.142	0.084	-1.686	0.093
		Residual standard error= 42.63;			*n* = 222
Soil CH_4_flux	*a*	1.618	4.028	0.402	0.688
	b	-1.421	7.809	-0.182	0.856
		Residual standard error= 40.27;			*n* = 222

4.4 Discussion

4.4.1 Influence of soil amendments on greenhouse gas fluxes during spring freeze-thaw

Nitrous oxide emission from compost amended soil was notably different from that of biosolids and digestate. This was likely due to the variation in carbon and nitrogen availability (see Table 3.2) and microbial activity among the different biobased residues evaluated in this chapter. Compost had a high N_2O flux compared to the other treatments, although this biobased residue has the lowest N mineralization potential (see Table 3.2). This possibly occurred because compost has the highest organic carbon and nitrogen, which takes the longest to fully mineralize, and the high moisture availability. Although, no soil moisture data except the visual observation, the compost-treated plot had the highest residual C and N in the next year's spring when the experiment was carried out. Similarly, Kariyapperuma et al. (2012) found also that the variable composition of composted swine influenced N_2O emissions during the non-growing season that undergoes spring freeze-thaw. Generally, composts with higher organic carbon and available nitrogen content cause greater N_2O fluxes. As the compost carbon/nitrogen ratio increases, N_2O fluxes decrease (Santos et al., 2021), suggesting the chemical composition of compost could regulate soil N_2O fluxes.

The lower-than-expected N_2O flux for anaerobic digestate in this chapter (Table 4.1) was due to the presence of readily degradable C and the physical quality (i.e., the liquid state of this residue type). These findings were also consistent with other studies that found lower N_2O fluxes with digested materials that were in a liquid form compared to either undigested materials (e.g., solid-state) or nitrogen fertilizer (Petersen 1999; Börjesson and Berglund, 2007; Chantigny et al. 2007; Collins et al. 2011; Rodhe et al., 2015; Baral et al., 2017). It has been shown that the low soluble organic C in anaerobic digestate leads to a lower source of energy for denitrifiers causing a reduction in the production of N_2O (Vallejo et al., 2006; Nkoa, 2014). Furthermore, digestate contains high soluble nitrogen readily available for plant uptake (De Vries et al., 2012; Baral et al., 2017). Approximately 98% of the anaerobic digestate used in this research was in the mineralized form and was probably taken up by the maize crop planted in the 2020 growing season (see Table 3.2).

Some previous studies showed that during biosolids production and throughout the first month of its application to agricultural soil, volatile nitrogen (e.g., ammonia) is lost from the soil (e.g., Yoshida et al., 2015; Roman-Perez et al., 2021). In this chapter, volatilization to ammonia might likely cause the lower N_2O flux observed in the biosolids treatment (Table 4.1). Furthermore, the biosolids used in this research was obtained from Lystek, which uses a thermal hydrolysis process that disintegrates the cell walls of microbes and hydrolyses complex macromolecules into simpler compounds (Lystek, 2021). The use of heat, shearing, and alkalinity during thermal hydrolysis can cause the cell walls to lyse, and it reduces the abundance of denitrifying bacteria, while carbon substrates persist and contribute to CO_2 and/or CH_4 emissions (Knowles, 1982; Murray et al., 2019). Results from this chapter also showed that the N mineralization rate of the Lystek biosolids was ~50% and indicated that most of the available N was taken up by the maize during the growing season (see Table 3.2).

A positive relationship was found between N_2O versus CO_2 and CH_4 emission among treatments which indicated that carbon input from biobased residues and nitrogen fertilizer stimulated the production of N_2O (Table. 4.2). Faubert et al. (2019) also found a positive relationship between N_2O and CO_2 emission and suggested that carbon input stimulated N_2O emission either by reducing soil oxygen availability through increased soil respiration or by the direct use of carbon from biobased residues as a source of energy for heterotrophic denitrifiers (Wrage-Mönnig et al., 2018). Since biobased residues can be distinctly different due to variation in their feedstock and/or production process, the quality of carbon (e.g., lignin content) and its nitrogen mineralization rate also varies (Ejack and Whalen, 2021). However, Roman-Perez et al. (2021) found that regardless of biobased residue type, the processes of carbon mineralization are similar, whereby increased C availability enhances microbial respiration and raises CO_2 and CH_4 fluxes (Roman-Perez et al., 2021). Furthermore, nitrogen mineralization in this research likely increased available nitrogen which was transformed to N_2O via nitrification or incomplete denitrification (Grant et al., 2020).

Although the cumulative CO_2 and CH_4 emissions and the global warming potential did not differ among treatments ($p>0.05$), a wide range of values was observed among the treatments (Table 4.1). This was due to differences in nutrient availability, especially the availability of carbon during the spring freeze-thaw, among treatments (Adviento-Borbe et al., 2007, Brenzinger et al., 2018). These results also supported the growing view among researchers that treatment comparisons should be explained using variability and magnitude effects and not only be limited to statistical significances (Wassertein et al., 2019, King et al., 2021). Furthermore, this chapter showed variation in greenhouse gas fluxes among treatments during the spring freeze-thaw, highlighting the importance of incorporating negative greenhouse gas fluxes. Since most studies in temperate environments do not account for greenhouse gas fluxes during winter and early spring (Yan et al., 2016; Kim et al., 2019; Natali et al., 2019). This chapter showed the importance of accounting for negative fluxes, especially in fertilized agroecosystems, since soil produces and consumes greenhouse gases (Hung et al., 2021).

4.4.2 Freeze-thaw phases and temperature influence on greenhouse gas fluxes

Soil freezing and thawing combined with changes in air and soil temperature caused a high degree of variability in CO_2 fluxes in all treatments. This indicated that soil warming regulated and enhanced microbial respiration during the spring freeze-thaw (Rafat et al., 2021; King et al., 2021; Byun et al., 2021). In this chapter, the CO_2 fluxes were different among freeze-thaw phases, where the dry phase had the highest flux compared to the wet and waterlogged phases. This was due to the variation in soil microbial respiration, which was enhanced when the soil became warmer (Ganjurjav et al., 2016; Gao et al., 2020). Furthermore, soil temperature was found to be lowest during the waterlogged phase (~ -0°C) compared to the wet (~ 2°C) and dry (~ 7°C) phases. Natali et al. (2019) also found that warming during freeze-thaw allowed for the short-term mineralization of organic residues, which caused the release of CO_2.

In addition, the intensity of soil respiration during freeze-thaw is positively dependent on soil temperature and moisture (Schipper et al., 2014; Byun et al., 2021). Since no crops or plants were present over the winter, the CO_2 flux quantified in this chapter directly measured soil microbial activity (Hung et al., 2021). Although moisture was not measured, it was observed that the significant relationship between temperature and CO_2 flux was a measure of the intensity of microbial metabolism even when the temperature dropped close to zero (-1 to 2°C) during the waterlogged and wet phase (Schaefer and Jafarov, 2016). This is because temperature controls a range of biogeochemical processes that regulate soil carbon cycling (Flanagan et al., 2013). Yan et al. (2016) found a significant and positive relationship between soil CO_2 flux and soil temperature during the spring freeze-thaw in a temperate forest ecosystem. Furthermore, Zou et al. (2018) summarized that multiple field studies found a significant and positive relationship between soil respiration intensity and soil temperature, and soil moisture. However, this chapter quantifies the relationship between CO_2 flux and soil temperature from various biobased residues during the spring freeze-thaw under field conditions.

Soil freezing and thawing caused a high degree of variability in N_2O and CH_4 fluxes indicating anaerobiosis caused by the soil moisture state of the phases contributed to the variability of the fluxes. The freeze-thaw phases in this chapter were determined with the visual moisture state, and the N_2O flux during the wet phase was significantly highest compared to the waterlogged and dry phases. This was likely due to the release of trapped N_2O in soil pores during the wet phase. The waterlogged phase causes soil pores with transient liquid water and ice to enable microbes to undergo denitrification that produces N_2O (Kim et al., 2012; Congreves et al., 2018). The N_2O produced could however be trapped because of high tortuosity created by water and ice in the pores (Hung et al., 2021, King et al., 2021) and gradually released when the waterlogged phase transitioned to the wet phase, where the ice seal was removed. In addition, CH_4 flux during the wet phase was different and higher compared to dry but not during the waterlogged phase. This suggests that the anaerobic and wet conditions encourage microbial production of CH_4 through methanogenesis (Luo et al., 2013). Further research should consider in-depth evaluation of the simultaneous contribution of spatial and temporal soil temperature and soil moisture measurements on NO_2 and CH_4 fluxes during spring freeze-thaw in temperate agricultural land.

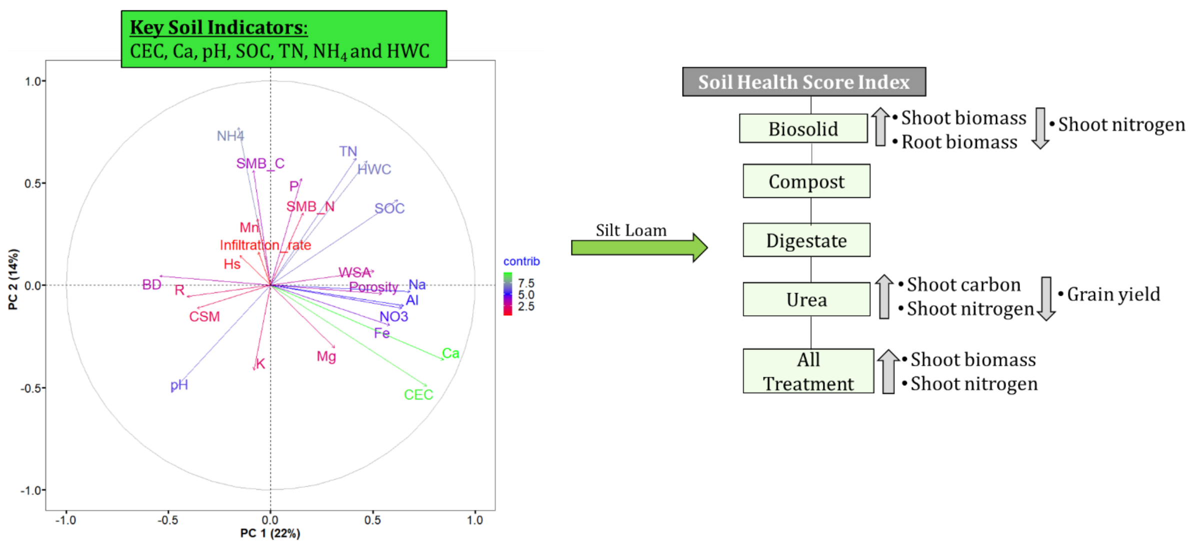
4.5 Conclusions

The impact of biobased residues on greenhouse gas fluxes during the spring freeze-thaw was evaluated. Greenhouse gas fluxes from biobased residues are not different compared to nitrogen fertilizer except for N_2O emission from compost. There was a significant relationship between soil temperature and CO_2 flux but no relationship with N_2O and CH_4 fluxes during the spring freeze-thaw. Among the freeze-thaw phases, the greenhouse gas fluxes were different. The dry phase during freeze-thaw caused intensified CO_2 fluxes compared to the wet and waterlogged freeze-thaw phase due to enhanced warming. Greenhouse gas fluxes from treatments with biobased residues and nitrogen fertilizer were controlled by the availability of carbon and nitrogen that regulated microbial processes of carbon mineralization and nitrification/denitrification. Future studies should incorporate molecular techniques to elucidate the specific organisms responsible for carbon and nitrogen transformations during spring freeze-thaw events, especially in the waterlogged and drying phases. This information will help further understand the role of biobased residues versus nitrogen fertilizers in greenhouse gas fluxes and aid with management decisions concerning the type and timing of biobased residue application.

Chapter 5

Impact of biobased residues on soil health in a temperate agricultural field

Graphical Abstract



5.1. Introduction

The mineral fertilizer industry is currently confronted with supply-chain shortages and rising production costs. In 2021, the cost of nitrogen fertilizers rose by 300% due to the high price of natural gas and carbon pricing (DTN, 2022; Tesio et al., 2022). The continuation of rising fertilizer costs may further contribute to global food security issues (Chojnacka et al., 2022). Conversely, disposal of organic materials including food waste, leaf and yard waste, and wastewater biosolids require large tracts of land to permanently store these waste materials (herein referred to biobased residues). However, biobased residues may be a more cost effective and environmentally sound replacement for mineral fertilizers (Badewa and Oelbermann, 2020, Cristina et al., 2020). This is because biobased residues are readily accessible since they are derived from inexpensive locally recycled materials (Chojnacka et al., 2022). Furthermore, biobased residues are high in bioavailable nutrients and can therefore support soil health and crop productivity (Vanotti et al., 2019; Chojnacka et al., 2022). Consequently, an assessment of the impact of different types of biobased residues on soil health and crop productivity is needed since it could help address fertilizer and food security challenges (Badewa and Oelbermann, 2020).

Soil health is viewed as an overarching concept critical for sustainable agriculture because it integrates different components including soil properties, processes, and interactions (Lehmann et al., 2020; Baveye, 2021) that are also be influenced by agricultural management practices (Doran and Zeiss, 2000; Lal, 2011). Soil health is defined as "the continued capacity of soil to function as a vital living system, with ecosystem and land-use boundaries, to sustain biological productivity, maintain the quality of air and water environments, and promote plant, animal, and human health" (Doran et al., 1996). However, the quantification of soil health requires several indicators to be assessed and integrated into an index (Lehmann et al., 2020). These indicators are referred to as physical, chemical, and biological soil properties and processes that can be readily measured to monitor changes in soil functions and sensitivity to management (Stott, 2019, Lehmann et al., 2020). According to a review by Bünemann et al. (2018) that included 65 studies, the frequency of incorporation of physical soil indicators was between 15% to 60%, chemical indicators; 15% to 90%, and biological indicators; 15% to 30%. In addition, there are different soil health assessments that have integrated different soil indicators to generate index scores. For example, these assessments include the soil management assessment framework (SMAF), the Haney soil health test (HSHT), the Soil Health Institute (SHI), and the comprehensive assessment of soil health (CASH). However, due to variations in the assessment methods of the various soil health indicators and the need for short-term responsive indicators, the integration and review of new soil health indicators are essential (Stott, 2019, Toor et al., 2021). Stott (2019) noted that standardized field and laboratory protocols for sampling and analysis, in addition to a 3 to 5-year review and update of indicators, are crucial to understanding the soil's health status. In addition, to fully understand soil health at a regional and site-specific level, there is a need to modify existing soil health assessments (Congreves et al., 2015).

Biobased residues are organic amendments produced by passing organic waste through the biobased production chain (Ho et al., 2017, Badewa and Oelbermann, 2020). Biobased residues such as compost, biosolids, and anaerobic digestate could therefore play an important role to help maintain or improve soil health (Chen et al., 2018). For example, organic amendments including biobased residues are reported to positively affect soil health indicators and processes, including water availability, nutrient availability, and nutrient cycling (Chen et al., 2018; Epelde et al., 2018; Verheijen et al., 2019). Although numerous studies evaluated the benefits of biobased residues on soil and its properties (Wu et al., 2013; Chen et al., 2018; Epelde et al., 2018), only a few studies conducted a comprehensive evaluation that includes a broad selection of physical, chemical, and biological soil properties (Congreves et al., 2015, Zhang et al., 2020). Since the application of biobased residues to agricultural land is expected to increase, it is imperative to compare the results of various types of biobased residues to that of nitrogen fertilizer to evaluate their impact on soil health.

Therefore, the objectives of this study were to determine (1) the impact of different biobased residue types and growing season on soil health and crop productivity, (2) if a relationship exists between soil health indicators and crop yield for different types of biobased residues and, (3) to contribute to the currently narrow database of biobased residue types on soil health. I hypothesized that there will be a greater improvement in soil health and crop productivity over the growing season with biobased residues compared to nitrogen fertilizer due to the higher input of organic matter and nutrients from biobased residues, and that soil health indicators and a soil health assessment score are correlated to crop yield since they inform soil health.

5.2. Materials and Method

5.2.1. Experimental site description and design

See Chapter 1, section *1.*3 for a description of the research location for experimental site description and design. See Table 1.1. for other detailed management at the location prior and during the experiment. The site was under maize (*Zea mays* L.) -soybean (*Glycine max* L.) -maize rotation.

5.2.2 Soil sampling and properties

Soil was randomly sampled at two depths (0-10 cm and 10-20 cm) from 3 random points in each treatment replicate. Soils were sampled before the start of the experiment to provide a baseline and in each year after crop harvest. Soil bulk density and porosity were measured according to McKenzie et al. (2002). Water stable aggregates were determined using a modified protocol from Carter et al. (2002) and Mehuys et al. (2007). Infiltration rate was measured using a 2800 Guelph Permeameter (Eijkelkamp

Agrisearch Equipment, Giesbeek, the Netherlands). Soil pH was determined using a 1:2 soil-water ratio using a pH meter (Fisherbrand, Accumet). Soil inorganic nitrogen (ammonium and nitrate), and available phosphorus (orthophosphate) were determined by colorimetry (Maynard and Kalra, 1993; Kuo, 1996). Soil organic carbon and total nitrogen were analyzed using an elemental analyzer (Costech ECS 4010). Soil organic matter was calculated by multiplying Soil organic carbon (%) by 1.72 (Jobbágy and Jackson, 2000). Soil cation exchange capacity, macro- and micronutrients were measured using the Atomic Absorption and Flame Photometry and Inductivity Coupled Plasma Atomic Emission Spectrometry (ICP-AES) after acidic digestion (Carson, 1980; Jones et al., 1991). Hot-water extractable carbon was adapted from Ghani et al (2003).

Soil microbial biomass carbon and nitrogen were determined using chloroform fumigation extraction (Voroney et al., 2008). Soil microbial community structure (microbial carbon substrate utilization) was determined using Biolog EcoPlates™ (Garland and Mills, 1991). Ecoplate readings were taken twice per day using a microplate reader (BioTek EL 800). Average well colour development was calculated to normalize the optical density values at maximum peak colour development on day 7 (Perujo et al., 2020). The richness of species and Shannon Diversity index were calculated according to the equation described by Shannon and Weaver (1949).

5.2.3 Crop Yield and Plant biomass

Crop biomass (root, shoot, and grain yield) were determined in October of each year. Each plant within a 2 m long, 0.4 m wide, and 0.2 m deep subplot was harvested. The subplots were located randomly within each treatment replicate. Maize and soybean plants were separated into above and below-ground components removing pods and cobs. Crop roots were washed over a 2 mm sieve to capture fine roots during the soil removal process (Böhm, 1979). All plant samples were oven-dried at 72°C for 3 d. The roots and a 50-50 mixture of stems and leaves ground in a Kinematica Polymix plant grinder (Px-MFC 90D, Kinematica, Lucerne, Switzerland), followed by grinding in a Retsch ball mill. The ground samples were analyzed for carbon and total nitrogen (Costech 4010, Valencia, CA, USA).

5.2.4 Soil Health Assessment

Soil health was assessed by incorporating soil physical, chemical, and biological indicators that also impact crop productivity (Figure 5.1). The selected soil indicators were responsive to short-term changes in management, measurable, and related to soil functions and ecosystem processes (Bünemann et al., 2018). The overall soil health score was developed using the weighted mean approach that integrates individual soil health scores (Congreves et al., 2015). Principal component analysis was used to construct weighting factors by assessing trends and variable correlations in all datasets (Wu and Congreves, 2021). Data were grouped based on treatment, depth, and year. The weighting factors were calculated using the sum of the first four component dimensions selected based on the inflection point from the Scree plot, with a percentage of explained variance (>5%).

The soil health score was computed using equation 5.1,

$$Soil\ Health\ Score\ = \frac{\sum_1^n(A_n \text{ x } A_n)}{\sum_1^n(w_n)} \tag{5.1}$$

Where A represents the soil health score (0-100) for each individual soil variable; w is the corresponding weighting factor. The soil health score was normalized from 0-100 to assess the management impacts on soil health, with a higher score expressing better soil health status.

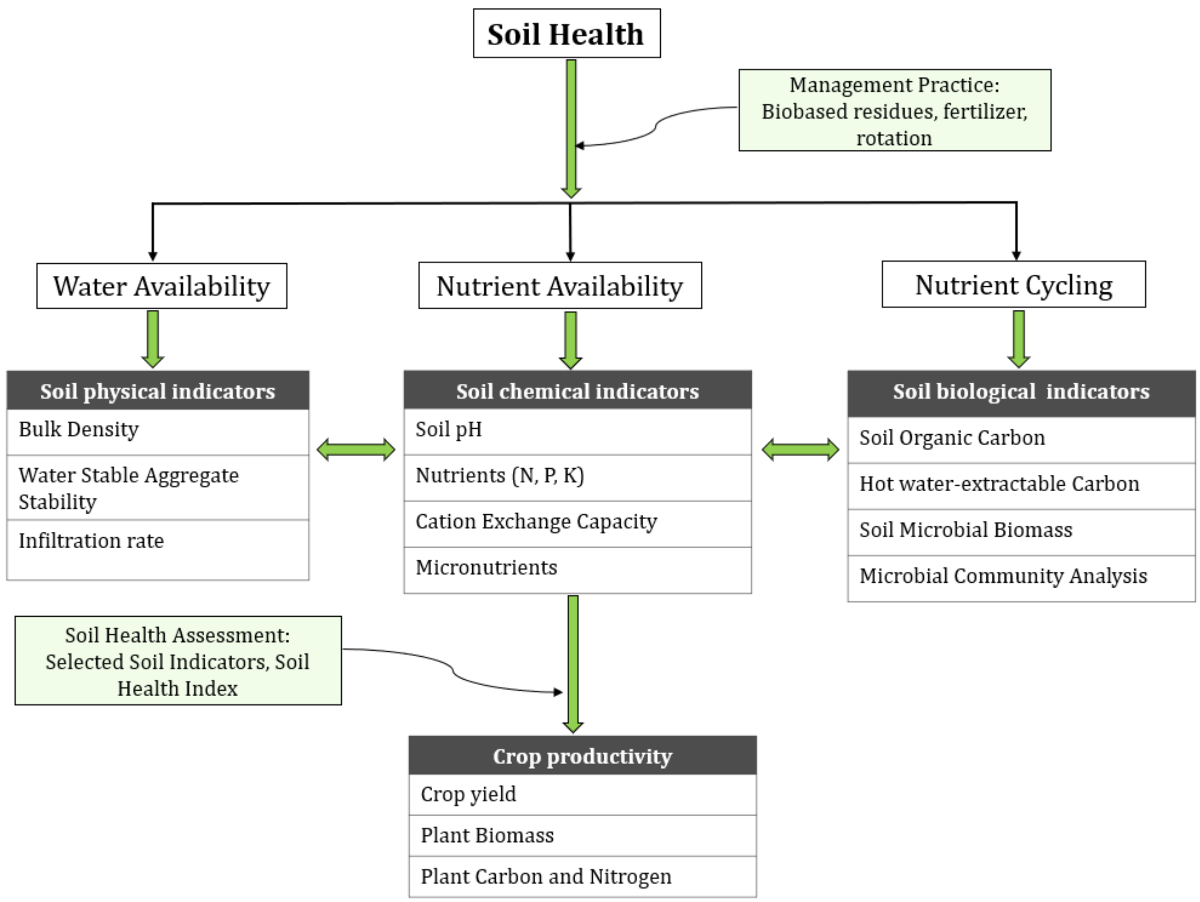


Figure 5.1: Soil health indicators used in this assessment suggested by the Soil Health Institute.

5.2.5 Statistical Analysis

All data processing and analysis were carried out in R *v.4.2.1* (R Core Team, 2020). Normality of soil indicators was determined using the Shapiro-wilt test, and box cox transformation was carried out when normality failed. Parametric statistics was employed after normality and equal variance criteria was met. A three-way analysis of variance was carried out for the data with the factors treatment, year, depth, and their interactions. Analysis with a significant F value (p<0.05) for the treatments was further compared using the Tukey test. Principal component analysis was performed using the FactoMineR package in R. Pearson's correlation coefficients were determined for soil and crop productivity indicators.

5.3. Results

5.3.1. Soil properties

Water stable aggregates, nitrate, available phosphorus, magnesium, potassium, sodium, and hot-water extractable carbon were significantly influenced by treatment, year, and depth (p<0.05; Table S5.1). Overall, water-stable aggregates were reduced by biobased residues, although compost and biosolids were not significantly different compared to nitrogen fertilizer (p>0.05; Table 5.1). Nitrate was not significantly different between the nitrogen fertilizer and biobased residues except for compost (p<0.05; Table 5.1). Available phosphorus was lowest for digestate and highest for biosolids, although not significantly different ($p>0.05$; Table 5.1). Specifically, biosolids had significantly higher available phosphorus compared to other treatments at 0-10 cm ($p<0.05$; Table S5.2) but was not significantly different at 10-20 cm (Table S5.2). Magnesium in the digestate treatment was lower and significantly different from nitrogen fertilizer ($p<0.05$) but not different ($p>0.05$) for compost and biosolids (Table 5.1). Potassium and sodium were increased by biobased residues and significantly different from nitrogen fertilizer ($p<0.05$) except for digestate (Table 5.1). Hot-water extractable carbon in the treatment with digestate was lowest and significantly different from nitrogen fertilizer treatment ($p<0.05$) (Table 5.1).

Soil microbial community structure changed after treatment application and shifted from the first year of the experiment (Year 1) to its final year (Year 3) (Figure 5.2). Generally, the microbial activity and utilization of carbon substrates in the treatments with biosolids and nitrogen fertilizer exhibited a longer distance between them compared to digestate and compost (Figure 5.2). In addition, utilization of carbon substrates was consistent throughout the evaluation for all treatments (Figure 5.2). However, the Shannon diversity index and microbial richness did not differ significantly among treatments (data not shown).

Table. 5.1: Soil indicators that were directly affected by nitrogen fertilizer and biobased residues at depth 0-20 cm in Elora Research Station, Ontario, Canada.

Treatment	Year	WSA (%)	NO_3^--N (mg kg^{-1})	PO_4^{3-}-P (mg kg^{-1})	Mg (cmol kg^{-1})	K (cmol kg^{-1})	Na (cmol kg^{-1})	HWC (mg kg^{-1})
Nitrogen Fertilizer	Year1	52.0 (3.9)	**22.0 (3.2)[a]**	12.4 (4.4)	4.3 (0.1)	0.6 (0.2)	**0.0 (0.0)[b]**	315.6 (35.9)
	Year2	24.7 (4.4)	7.2 (0.5)	12.9 (3.2)	3.6 (0.2)	0.5 (0.1)	0.0 (0.0)	265.4 (17.7)
	Year3	15.0 (2.8)	**6.7 (1.2)[ab]**	2.7 (1.4)	4.3 (0.2)	0.5 (0.1)	0.0 (0.0)	176.8 (13.0)
	Year (1-3)	**30.6 (3.9)[a]**	**12.0 (1.8)[a]**	9.3 (2.0)	**4.1 (0.1)[a]**	**0.5 (0.1)[b]**	**0.0 (0.0)[b]**	**252.6 (18.0)[a]**
Digestate	Year1	44.9 (9.0)	**17.4 (2.2)[ab]**	6.5 (1.2)	3.4 (0.5)	0.4 (0.1)	**0.1 (0.0)[ab]**	244.9 (19.3)
	Year2	16.7 (2.5)	6.7 (0.4)	7.3 (1.7)	3.3 (0.2)	0.5 (0.0)	0.0 (0.0)	250.9 (18.0)
	Year3	9.6 (1.0)	**12.3 (2.1)[a]**	2.5 (1.1)	3.9 (0.3)	3.1 (1.0)	0.0 (0.0)	155.6 (8.7)
	Year (1-3)	**23.8 (4.4)[b]**	**12.1 (1.3)[a]**	5.4 (0.9)	**3.6 (0.2)[b]**	**1.3 (0.4)[a]**	**0.0 (0.0)[ab]**	**217.1 (12.7)[b]**
Compost	Year1	49.6 (5.7)	**10.1 (2.1)[b]**	5.8 (1.4)	3.8 (0.1)	0.7 (0.4)	**0.2 (0.0)[a]**	273.1 (27.8)
	Year2	21.5 (2.6)	6.9 (0.3)	13.2 (3.8)	3.4 (0.2)	0.4 (0.0)	0.0 (0.0)	257.2 (15.4)
	Year3	10.6 (0.8)	**5.9 (0.6)[b]**	1.9 (0.8)	3.8 (0.2)	2.0 (1.0)	0.0 (0.0)	147.8 (17.9)
	Year (1-3)	**27.2 (4.0)[ab]**	**7.6 (0.8)[b]**	7.0 (1.6)	**3.6 (0.1)[ab]**	**1.0 (0.4)[ab]**	**0.1 (0.0)[a]**	**226.0 (16.4)[ab]**
Biosolids	Year1	45.5 (8.7)	**13.1 (2.4)[ab]**	8.1 (2.1)	4.0 (0.1)	0.5 (0.2)	**0.2 (0.1)[a]**	289.1 (27.3)
	Year2	21.8 (3.5)	6.5 (0.3)	13.1 (3.6)	3.5 (0.1)	0.4 (0.0)	0.0 (0.0)	251.3 (25.1)
	Year3	12.3 (1.0)	**8.0 (1.7)[ab]**	7.0 (3.1)	4.4 (0.1)	2.9 (0.9)	0.0 (0.0)	175.6 (15.4)
	Year (1-3)	**26.5 (4.2)[ab]**	**9.2 (1.1)[ab]**	9.4 (1.7)	**4.0 (0.1)[ab]**	**1.3 (0.4)[a]**	**0.1 (0.0)[a]**	**238.7 (16.1)[ab]**

Values are means with standard errors in bracket. Mean with significantly difference (p<0.05) among treatments within year are bolded and indicated with different letters. **Note:** WSA, water stable aggregates; NO_3^--N, nitrate; PO_4^{3-}-P, orthophosphate; Mg, magnesium, K, potassium; Na, sodium; HWC, hot water extractable carbon.

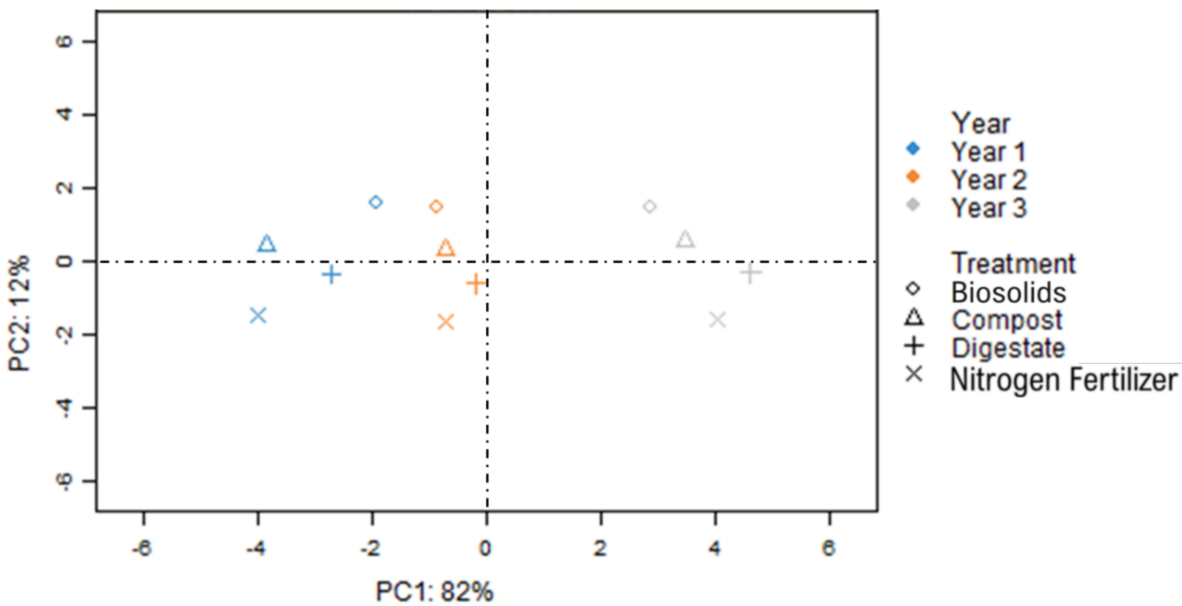


Figure 5.2: The principal component analysis presenting the microbial activity and utilization of carbon substrate among the treatments and years for the first two principal components at Elora Research Station, Ontario, Canada.

5.3.2. Soil Health Assessment

All selected soil indicators were either significantly positive or negative correlated with crop yield, plant biomass, or shoot carbon and nitrogen except for bulk density, porosity, infiltration rate, soil organic carbon, soil microbial biomass carbon and nitrogen, capacity of soil microbes to utilize carbon substrates, and microbial richness (p<0.05; Table 5.2). The cumulative percent variability accounted for by the first two components was 34%, thus describing most of the variation among the soil indicators (PC1 - 22% and PC2 - 14%) (Figure S5.1). Based on the factor pattern between the first two principal components (Figure 5.3), five major groups of related soil health indicators were identified (Table 5.3). The principal components were predominately explained by indicator type. For instance, principal component 1 was explained by physical and chemical indicators, where principal component 2 was by chemical and biological indicators. The indicators with the greatest weighting factors were soil organic carbon, total nitrogen, soil microbial biomass carbon and nitrogen, cation exchange capacity, hot water extractable carbon and manganese (Figure S5.2).

Table 5.2: Correlation coefficients (r) between soil indicators and crop yield, plant biomass or shoot carbon and nitrogen at Elora Research Station, Ontario, Canada.

Soil Indicators	Grain Yield	Root Biomass	Shoot Biomass	Shoot Nitrogen	Shoot Carbon
BD (g cm^{-3})	0.09	-0.02	-0.22	0.17	-0.21
WSA (%)	**-0.41****	-0.23	0.04	0.00	-0.04
Porosity (%)	-0.09	0.02	0.22	-0.17	0.22
Infiltration rate (mm hr^{-1})	-0.09	-0.04	-0.08	0.16	-0.13
pH	**0.47*****	**0.37****	0.13	-0.22	-0.05
NO_3—N (mg kg^{-1})	0.01	0.12	**0.47*****	**-0.29***	0.28
NH_4^+-N (mg kg^{-1})	**-0.69*****	**-0.73*****	**-0.68*****	**0.67*****	-0.25
PO_4^{3}—P (mg kg^{-1})	**-0.45*****	**-0.44****	-0.21	0.28	-0.18
SOC (%)	-0.11	-0.01	0.28	-0.05	0.26
TN (%)	**-0.36***	**-0.28***	-0.01	0.23	0.18
CEC (cmol kg^{-1})	0.28	**0.49*****	**0.78*****	**-0.60*****	**0.38****
Ca (cmol kg^{-1})	0.27	**0.48*****	**0.76*****	**-0.61*****	**0.35***
Mg (cmol kg^{-1})	0.28	**0.30***	**0.43****	-0.28	**0.35***
K (cmol kg^{-1})	0.12	0.12	**0.32***	-0.14	0.26
Na (cmol kg^{-1})	0.06	**0.29***	**0.54*****	**-0.42****	**0.31***
Al (cmol kg^{-1})	0.10	**0.31***	**0.52*****	**-0.44****	-0.14
Mn (cmol kg^{-1})	-0.27	**-0.29***	-0.27	0.28	-0.07
Fe (cmol kg^{-1})	0.07	0.27	**0.47***	-0.16	0.10
HWC (mg kg^{-1})	**-0.48*****	**-0.33***	0.01	0.02	0.10
SMB-C (μg g^{-1})	-0.18	-0.19	-0.21	0.24	-0.09
SMB-N (μg g^{-1})	0.13	0.04	0.01	0.01	-0.01
CSM	0.54	0.31	-0.28	-0.15	-0.08
Hs	**-0.59***	**-0.72****	**-0.77****	0.57	0.21
R	0.20	-0.04	-0.57	0.14	-0.04

The values shown are r-values. The r values in bold text indicate that effects were significant at the 0.05 level or lower. ns: not significant. *, **, *** indicate $p<0.05$, $p<0.01$ and $p < 0.001$, respectively.

Table 5.3: Category of related soil health indicators at Elora Research Station, Ontario, Canada.

Group	Related soil health indicators	Category
Group 1	Water stable aggregates, porosity, sodium, aluminum, nitrate, iron, calcium, cation exchange capacity, magnesium	Physical and chemical
Group 2	Soil organic carbon, hot water extractable carbon, total nitrogen, soil microbial biomass nitrogen, available phosphorus	Chemical and biological
Group 3	Soil microbial biomass carbon, ammonium, manganese, infiltration rate and Shannon diversity index	Physical, chemical, and biological
Group 4	Bulk density, microbial richness, capacity of soil microbes to utilize carbon substrates	Physical and biological
Group 5	pH and potassium	Chemical

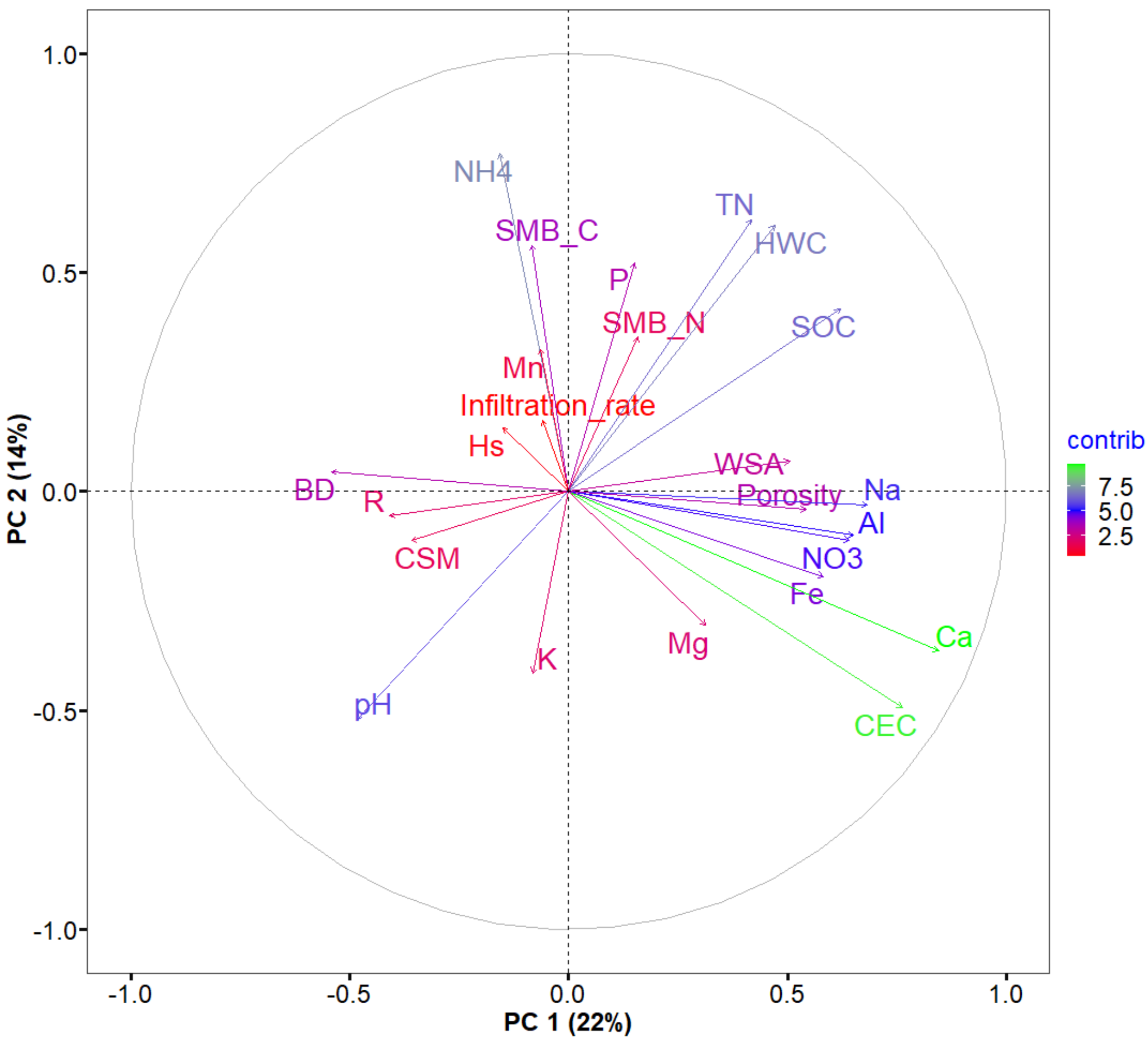


Figure 5.3: The principal component analysis of soil indicators at 0-20 cm depth under nitrogen fertilizer-biobased residues application at Elora Research Station, Ontario, Canada. The color of the soil indicators indicates their contribution (contrib.) to the first two principal components.

The first two principal components determined show a clear relationship between the components for the year and, to a lesser extent, for soil depth, but there was no clear relationship for the treatment effect (Figure 5.4). Overall, the soil health score was not significantly different among treatments. For example, biosolids had the highest score and compost the lowest (p>0.05; Figure 5.5). However, the soil health score for compost was significantly lower (p<0.05) compared to the other treatments in year 3. The soil health score for biosolids was also positively correlated with the shoot and root biomass and negatively correlated with shoot nitrogen (p<0.05, Table 5.4). Although, there was no significant correlation (p>0.05) between grain yield and shoot carbon for biosolids soil health score (Table 5.4). The soil health score for compost and digestate were not significantly correlated with crop productivity indicators (p>0.05) while only shoot nitrogen and carbon were positively correlated with the soil health score for nitrogen fertilizer (p<0.05) (Table 5.4).

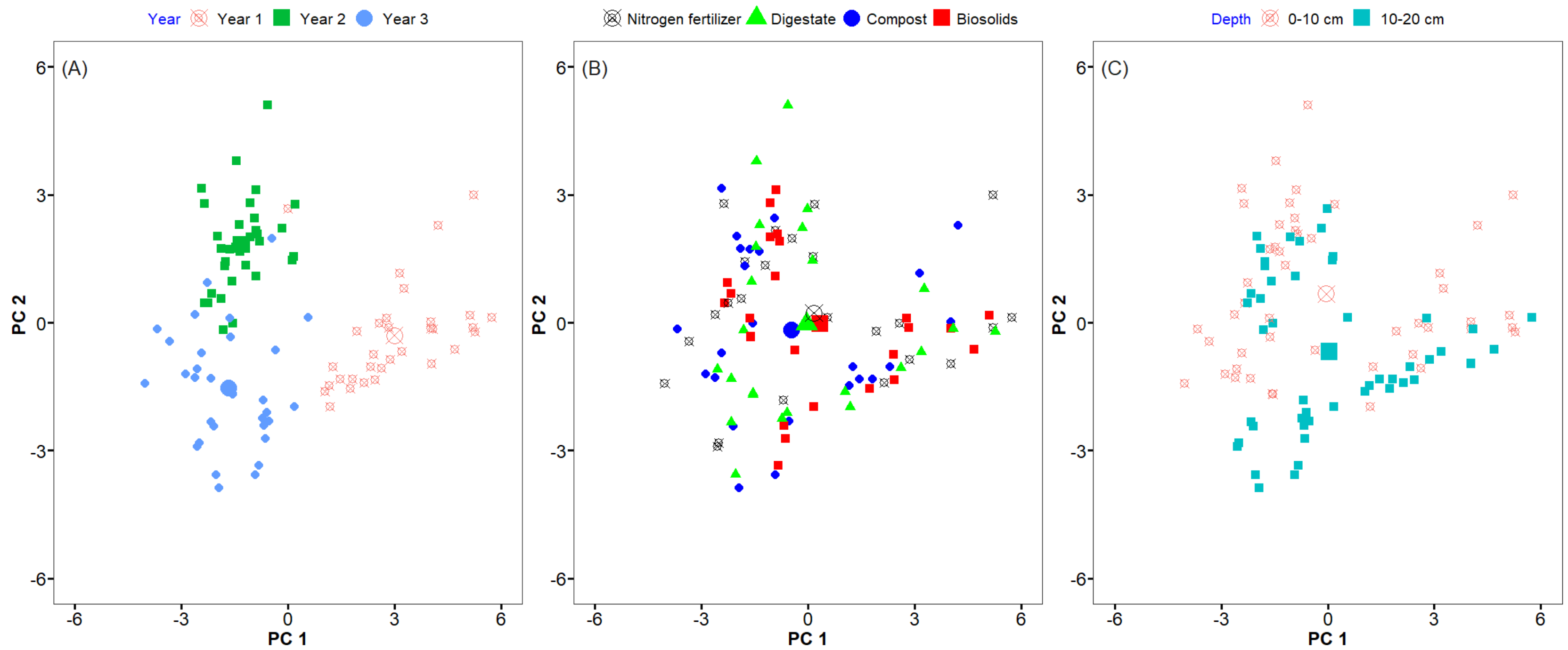


Figure 5.4: The principal component analysis (first two-components) of all soil indicator data observed to evaluate the effect of (A) Year, (B) Treatment, (C) Depth from the experimental site at Elora Research Station, Ontario, Canada.

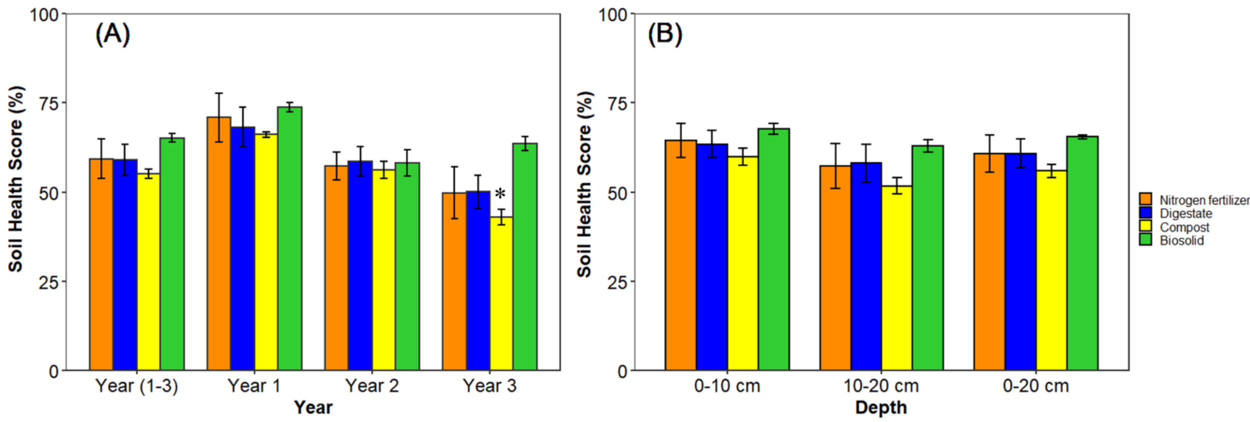


Figure 5.5: Soil health assessment scores with standard error for the nitrogen fertilizer-biobased residues treatment across the (A) year and (B) depth computed at Elora Research Station, Ontario, "*" denotes a significant biobased residues treatment effect (p<0.05).

Table 5.4: Correlation coefficients (r) between Soil Health Score and crop yield, plant biomass or shoot carbon and nitrogen at Elora Research Station, Ontario, Canada.

Treatment	Year	Grain Yield	Shoot Biomass	Root Biomass	Shoot Nitrogen	Shoot Carbon
Nitrogen Fertilizer	Year1	**-0.96***	-0.68	0.79	-0.67	0.90
	Year2	-0.54	-0.23	0.17	-0.03	-0.29
	Year3	-0.47	-0.37	-0.89	**0.98***	0.46
	Year (1-3)	-0.10	0.36	-0.08	-0.34	**0.73***
Digestate	Year1	0.16	-0.23	-0.38	0.87	-0.17
	Year2	0.22	0.68	-0.02	-0.02	0.82
	Year3	0.17	-0.17	-0.52	0.82	-0.71
	Year (1-3)	0.08	0.40	0.16	0.14	0.26
Compost	Year1	-0.01	0.56	-0.48	0.51	0.51
	Year2	-0.35	-0.74	-0.65	-0.06	0.75
	Year3	-0.06	0.02	-0.01	-0.54	0.29
	Year (1-3)	-0.44	0.45	0.08	-0.12	0.40
Biosolids	Year1	-0.05	-0.35	-0.15	-0.62	-0.01
	Year2	0.25	0.47	-0.03	-0.23	-0.61
	Year3	0.12	-0.62	-0.28	0.38	-0.39
	Year (1-3)	0.42	**0.76***	**0.62***	**-0.77***	0.04
Treatment	Year1	-0.08	-0.26	-0.04	0.23	0.06
	Year2	-0.04	0.08	-0.06	-0.12	-0.02
	Year3	-0.24	0.15	-0.50	**0.66***	0.07
	Year (1-3)	-0.06	**0.44***	0.11	-0.19	0.28

The values shown are r-values. The r values in bold text indicate significant relationship at the 0.05 level or lower. "*" indicate $p < 0.05$.

5.4. Discussion

5.4.1 Biobased residues and soil properties

Biobased residues have both positive and negative effects on soil properties. The variation of the influence of biobased residues, compared to nitrogen fertilizer, on soil properties indicated the importance of biobased residues' chemical composition. Generally, biobased residues increase soil fertility due to their relatively higher amount of macro and micronutrient content compared to nitrogen fertilizer. In this chapter, the nutrients supplied from biobased residues varied depending on their feedstock, production process, and carbon/nitrogen ratio (Table 3.1). A review by van Zwietan (2018) noted that adding organic amendment provides direct and long-term supply of macro and micronutrients regulated by the mineralization rate of the organic material. Nutrient content also depends on the source and quality of the biobased residues (Quilty and Cattle 2011).

However, biobased residues decrease water-stable aggregates and hot-water extractable carbon compared to nitrogen fertilizer. This was likely due to the relatively high sodium level in the biobased residues that disrupted soil aggregates and biobased residues' ability to enhance microbial biomass turnover. Panayiotopoulos and Leinas (2009) found that soil with high sodium concentration increased soil cracks and soil dispersion that decreased the stability of aggregates. Ghani et al. (2003) observed that the nitrogen content in soil amendments enhanced microbial biomass turnover by assimilating relatively high labile forms of carbon, thereby causing a depletion in hot-water extractable carbon. Thus, biobased residues' application and production methods for agricultural use should consider minimizing the sodium content to reduce aggregate dispersal. Since, the value of these soil properties consistently decreased yearly and in all the treatments, the research site and its history could not be overlooked. This is because the qualitative observations of the soil during the evaluation period showed soil cracks when dry and marshy like soil when wet. This indicates that the clay particles cause expansion and contraction of the soil resulting in soil aggregate breaking.

The treatments accessed different carbon substrates throughout the experiment, with distinct differences between nitrogen fertilizer and biosolids (Figure 5.2). This was probably due to the sensitivity of the soil microbial community and differences in the availability of carbon and nitrogen among the treatments (Allison et al., 2007, Zhang et al., 2020). Similarly, Luo et al. (2018) observed that carbon, nitrogen content, and C/N ratio for organic amendments affected microbial-related soil functions. Epelde et al. (2018) also found that carbon substrate utilization depended on the variety of ingredients in organic amendments. Therefore, biobased residue characteristics related to their carbon and nitrogen content influenced the bacterial community and the carbon sources they metabolized.

5.4.2 Biobased residues and soil health

Soil health assessment scores for biobased residues were not different compared to that of nitrogen fertilizer, although there was wide variation among the scores. This was likely due to the variation in labile carbon and available nitrogen among the treatments. Based on the production method and type, digestate and biosolids consist of easily degradable carbon and nitrogen that are available for crop use (Nkoa, 2014; Lystek, 2021). At the same time, compost has complex and recalcitrant organic carbon and immobilized nitrogen that takes a longer time to fully be available for crop uptake (Santos et al., 2021). The soil health score was developed by choosing soil carbon/nitrogen and nutrient-based metrics to establish linkages between soil health and crop yield to assess the influence of biobased residues on soil health and crop production (Figure 5.1). The approach generated an overall score using individual soil indicators and their weighted average generated through principal component analysis. The indicators that influenced the soil health score were carbon, nitrogen, and nutrient concentration. Similarly, Wu and Congreves (2021) adopted a framework emphasizing soil carbon, nitrogen, and total nutrient concentration, providing a promising link between soil health scores and crop yield. van Es and Karlen (2019) also observed that labile carbon and available nitrogen indices are crucial to integrating soil health and crop productivity. Additionally, the soil health assessment method was able to capture the soil's ability to support crop productivity under biobased residues. This was shown by the positive correlation between the soil health score and related crop parameters for biosolids, nitrogen fertilizer, and all treatments combined (Table 5.4). Wu and Congreves (2021) also observed a positive correlation between soil health scores and crop yield from 55 arable fields across Saskatchewan, Canada.

Most of the soil indicators conceptually selected for the soil health assessment and how it relates to crop productivity suggested the selected soil indicators for the biobased residues soil health assessment was sufficient. This reaffirmed that sensitivity to management changes of the integrated indicators is crucial for soil health assessment (Toor et al., 2021). In addition, pH, nitrate, cation exchange capacity, and nutrients are related strongly to at least one of the crop productivity parameters (Table 5.2). Likewise, Cai et al. (2019) observed pH and soil nutrient content were the soil indicators responsible for yield increase after manure application. Agegnehu et al. (2016) also found that cation exchange capacity improved crop yield under organic amendments and nitrogen fertilizer. Although there was no difference between crop yield of nitrogen fertilizer and biobased residues, the direct experimental test and a wider range of indicators used in this evaluation supported the effort to establish causal links between organic amendments and crop yields (Luo et al., 2018).

Five groups of related soil indicators were identified based on the principal component analysis, with a major contribution from cation exchange capacity, calcium, pH, ammonium, soil organic carbon, total nitrogen, and hot water extractable carbon (Figure 5.3). This result confirmed that soil cation and anion interaction with soil organic carbon and nitrogen contributed significantly to soil health. Congreves et al. (2015) noted that soil organic carbon has negatively charged sites that are available for the adsorption and exchange of soil cations, such as calcium ions, and cation exchange capacity is directly related to negatively charged soil particles and exchangeable cations. Several studies (*i.e.,* Lal, 2016; Wu and Congreves, 2021) also observed that soil organic carbon and nitrogen were key components of soil organic matter critical for soil ecosystem functioning such as water and nutrient availability, nutrient recycling, climatic regulation, and plant growth. In addition, the principal component analysis showed pH was grouped in the opposite plane with available phosphorus, soil organic carbon, total nitrogen, and hot water extractable carbon, soil microbial biomass nitrogen, and suggested that pH influenced the availability of nutrients (Van Eerd et al., 2014, Chowdhury et al., 2021). Kooijman et al. (2009) also found that pH related strongly with soil microbial biomass nitrogen.

The soil health assessment score varied among years and depth regardless of the treatment, indicating that experiment duration and depth influence soil health indicators The principal component analysis grouping for year and depth was distinct (Figure 5.6) because selected soil indicators were responsive over the short term and tend to vary among depths. Congreves et al. (2015) also observed clear principal component analysis groups under tillage and crop rotation due to experiment duration influence on soil quality properties. Similarly, variation in soil health scores due to depth was observed in arable cropping systems in Saskatchewan (Wu and Congreves, 2021).

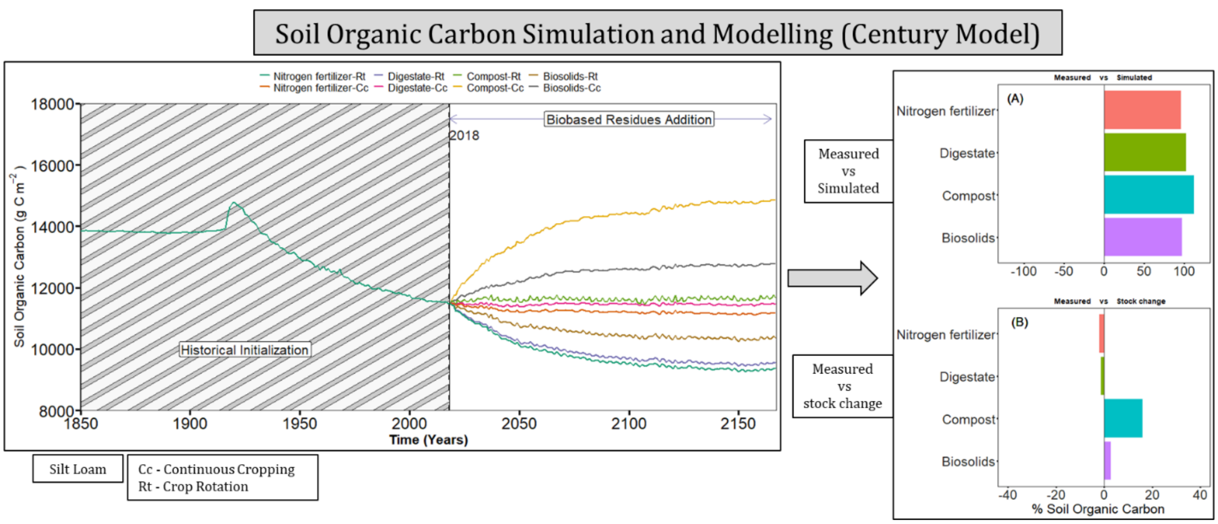
5.5. Conclusions

The impact of biobased residues on soil health in silt loam soil in southern Ontario was evaluated. Five related soil attributes were identified based on the principal component analysis. Biobased residues increased soil fertility but decreased water-stable aggregates and hot-water extractable carbon compared to nitrogen fertilizer. Generally, soil health assessment was different among the years and depths but not among treatments. Soil health score integrated the relationship among the physical, chemical, and biological soil indicators. Future studies should consider applying the soil health score approach for more comprehensive soil types and biobased residues. Classification of the impact of biobased residues on microbial community structure and diversity should be expanded using genetic sequencing and lipid analysis.

Chapter 6

The Century Model evaluates the long-term effects of biobased residues on soil organic carbon dynamics.

Graphical Abstract



6.1 Introduction

A reduction in soil organic carbon and an increase in global atmospheric carbon occur when undisturbed ecosystems, including forests and grasslands, are converted to intensively managed agroecosystems (Sanderman et al. 2017). However, agroecosystems are also capable of carbon sequestration and can have a significant impact on offsetting global greenhouse gas emissions (Lal et al. 2007; IPCC 2014). This has generated an international effort to establish programs (e.g., 4 per 1000 initiative www.4p1000.org) to determine the best agroecosystem management strategies to maximize carbon sequestration (Dimassi et al. 2018). For example, soil carbon sequestration can be achieved through conservation agriculture that minimizes mechanical soil disturbance (e.g., no-till), allows for a minimum of 30% soil cover (e.g., crop residue management, cover crops) and crop diversification (e.g., crop rotation) (Lal 2004; Smith 2004; Aertsens et al. 2013; Chenu et al. 2019).

A continuously growing global population and its increasing demand for food, fuel, feed, and fibre require the need for recycling residues from urban organic waste chains to agricultural land (Ho et al., 2017). Disposal of organic materials, including food waste, leaf and yard waste, and wastewater biosolids, into landfills, is associated with multiple environmental issues. For example, large tracts of land that could otherwise be used for food, fuel, feed, or fibre production, are required to permanently store these waste materials (herein referred to as biobased residues). The decomposition of biobased residues also contributes to air and water contamination and the emission of methane (Levis and Barlaz, 2011). Biobased residues, however, can be diverted and processed to produce carbon - and nutrient-rich soil amendment that improves soil's physical, chemical, and biological properties and crop productivity (van Zwieten, 2018). This may also provide an opportunity to decrease the rate of nitrogen fertilizer application to agricultural land and thereby mitigate nitrous oxide emissions. Thus, biobased residues have a significant economic and environmental value in the agricultural sector (Badewa and Oelbermann 2020).

Paustian et al. (2016) suggested that biobased residues are one of the proximal controls on soil carbon dynamics. This is because biobased residues contain a sufficient quantity of organic matter to help increase soil organic carbon stocks (Tian et al. 2009; Cogger et al. 2013; Nkoa 2014; Jin et al. 2015; Garvey et al. 2016). Although Stavi and Lal (2013) and Aertsens et al. (2013) suggested that the potential for soil carbon sequestration and stabilization may be most effective in conservation agriculture. Some studies suggested that conservation practices like cereal-legume crop rotations may not effectively increase soil organic carbon (West and Post, 2002; Govaerts et al., 2009). However, these studies did not evaluate the impact on soil organic carbon when crop rotation was combined with biobased residues. Furthermore, Dynarski et al. (2020) noted that currently, a lack of knowledge exists on the longevity and permanence of carbon in soil from the organic matter supplied by biobased residues in conventional or conservation agroecosystem management practices. This knowledge gap can be addressed with the aid of process-based models (e.g., Century Soil Organic Matter Model, DNDC, RothC) that can help predict changes and the rate of change in soil organic carbon stocks under continuous cropping and crop rotation (Oelbermann et al. 2017; Dimassi et al. 2018; Singh and Benbi 2020). Dimassi et al. (2018) used the Century model on six long-term experimental sites with 25 treatments that integrated different crop rotations with inorganic and organic amendments. They found that quantifying the change in the soil organic carbon stock avoided a mismatch between simulated and measured soil organic carbon levels caused by the Century model uncertainty and simulated systems characteristics, *e.g.,* historical initialization. The goal of this chapter was to evaluate how biobased residues, compared to nitrogen fertilizer, impact soil organic carbon stock and its associated fractions (active, slow, and passive) under continuous cropping and crop rotation in a temperate medium-textured soil in Ontario, Canada, using the Century model. It was hypothesized that i) biobased residues will increase soil organic carbon stocks and associated fractions under continuous cropping and crop rotation, due to their high carbon supply and that ii) validation using the change in soil organic carbon stocks will improve the century model performance compared to the simulated soil organic carbon due to its capacity to reduce century model uncertainties caused by historical initialization.

6.2 Materials and Methods

6.2.1 Research Site

See Chapter 1, section *1.3 Research location* for experimental site description, soil, and climate information. However, specific data input for century model simulation for soil parameters used was presented in Table S6.1, climate data; Table S6.2, and land management practices; Table 6.1.

Table 6.1: Scheduling of agricultural management practices in Century at Elora Research Station, Ontario, Canada.

Date	Management practices/Century Parameters
10,000 BC – 1914	Mixed deciduous/coniferous forest (MIX) ~10,000 years to estimate equilibrium soil organic matter levels and plants productivity and clear-cut harvesting
1915 – 2017)	Elora agricultural management practices – 100 years. <u>1915 to 1966</u>: grass/hay [100 % cool (GCD)]; hay (H) harvest. <u>1967 to 2017</u>: continuous maize (CHI-high harvest maize); plowing (P); grain only harvest (G); Nitrogen fertilizer (17 g N $m^{-2}y^{-1}$, 2.7 g P $m^{-2}y^{-1}$).
2018 - 2167	*Treatments* <u>*Nitrogen fertilizer under continuous cropping and crop rotation*</u> Nitrogen fertilizer-Cc: Continuous maize (CHI high harvest maize) Plowing (P); Nitrogen fertilizer added at 17 g N $m^{-2}y^{-1}$; grain only harvest (G) Nitrogen fertilizer-Rt: (CHI high harvest maize) – soybean (SYBN) - Winter wheat (SWHI) rotation Plowing (P); Nitrogen fertilizer added at 17 g N $m^{-2}y^{-1}$; grain only harvest (G) <u>*Biobased residues under continuous cropping and crop rotation*</u> Compost-Cc: Continuous maize (CHI high harvest maize) Plowing (P); compost added at 1200 g m^{-2} $2y^{-1}$; grain only harvest (G) Biosolids-Cc: Continuous maize (CHI high harvest maize) Plowing (P); biosolids added at 2800 g m^{-2} $2y^{-1}$; grain only harvest (G) Digestate-Cc: Continuous maize (CHI high harvest maize) Plowing (P); digestate added at 4200 g m^{-2} $2y^{-1}$; grain only harvest (G) Compost-Rt: maize (CHI high harvest maize) – soybean (SYBN) - Winter wheat (SWHI) rotation Plowing (P); compost added at 1200 g m^{-2} $2y^{-1}$; grain only harvest (G) Biosolids-Rt: maize (CHI high harvest maize) – soybean (SYBN)- Winter wheat (SWHI) rotation Plowing (P); biosolids added at 2800 g m^{-2} $2y^{-1}$; grain only harvest (G) Digestate-Rt: maize (CHI high harvest maize) – soybean (SYBN)-Winter wheat (SWHI) rotation Plowing (P); digestate added at 4200 g m^{-2} $2y^{-1}$; grain only harvest (G)

6.2.2 The Century model and model calibration

The Century model simulates changes in soil organic carbon and its associated fractions based on site-specific soil-plant-climate parameters (Smith et al. 2007). The microbial decomposition of plant residues and resulting microbial products, which are the basis for humus formation, are predicted using soil active, slow, and passive fractions in Century's soil organic matter sub-model (Parton et al. 1988). The active fraction, which includes soil microbes and microbial products, has a short turnover time (1-3 months) and is composed of about 2 to 4% of the total soil organic matter (Parton et al. 1987; Metherell et al. 1993). The slow fraction comprises 45 to 65% of the total soil organic matter pool with a turnover time of 10 to 50 years, depending on climate, and includes resistant plant material derived from structural plant material and stabilized soil microbial products (Metherell et al. 1993). The passive fraction comprises 45 to 50% of the total soil organic matter involved in the physical and chemical stabilization of soil organic matter. This fraction is resistant to decomposition and has a turnover time of 400 to 4000 years (Parton et al. 2001). The turnover time of the carbon fractions, soil temperature, and soil moisture determine the turnover time of soil organic carbon (Parton et al. 1987; Parton and Rasmussen 1994). Although Century was initially developed to simulate changes in grasslands, the model is capable of simulating changes in soil organic carbon in temperate and tropical agroecosystems and has also been applied to complex agroecosystems such as agroforestry (Oelbermann and Voroney 2011) and cereal-legume intercrops (Oelbermann et al. 2017).

Thus, Century has a broad applicability which has made it one of the most widely used models to evaluate long-term changes in soil organic carbon stocks (Oelbermann et al. 2017).

The Century model (version 4.0) was used to simulate long-term changes in soil organic carbon and its associated active, slow, and passive fractions at the biobased residue at the Elora Research Station in Southern Ontario, Canada. The monthly average maximum and minimum air temperature and monthly total precipitation data were obtained from an on-site meteorological station (Table S6.2). To calibrate the model, soil parameters collected from the experimental plots, before adding biobased residues or nitrogen fertilizer were used (Table S6.1). The proportion of soil organic matter initial values used for the microbial active, slow, and passive fractions was 2167, 3623 and 1784 g m^{-2} respectively. The maximum decomposition rate of soil organic matter values with active, slow, and intermediate turnover was 4, 0.0045 and 0.20 respectively.

Additional files were created in OMAD.100 and FERT.100 representing biobased residues input and the corresponding nutrient content of the biobased residues (Table 6.1). The addition of biobased residues was equivalent to a carbon input of 240.8 g C m^{-2} y^{-1} (compost), 105.6 g C m^{-2} y^{-1} (biosolids), 19.4 g C m^{-2} y^{-1} (digestate). The carbon/nitrogen ratio was 9 (compost), 6 (biosolids) and 0.4 (digestate). The FERT.100 file corresponded with nitrogen fertilizer input of 17g N m^{-2} y^{-1}. For the simulation, default values set by Century remained unchanged.

To fully understand the long-term effects of biobased residues on soil organic carbon dynamics, it is critical to initialize soil organic carbon fraction size for the Century model. This involves running the model iteratively for thousands of years by using the TREE file so as to establish an equilibrium with the initial soil organic carbon at the start of the different treatments. The initialization was based on a mixed deciduous-coniferous forest under temperate conditions from ~10,000 years to 1914. Followed by subsequent agricultural management practices that began in 1915 with 52 years of grass/hay, followed by 51 years of continuous maize from 1967 to 2017. After which, the treatments were initiated in 2018 and simulated for 150 years until 2167 (Table 6.1).

6.2.3 Soil Sampling

Soil samples were collected from each treatment replicate and analyzed for bulk density, soil organic carbon and total nitrogen at depth 0 -10 cm and 10 - 20 cm. Bulk Density was collected by inserting in the soil a ring (inner diameter: 4.5 cm, height: 5.1 cm). The soil inside the rings was then dried in an oven for 48 hours at 105 °C. Bulk density values were obtained by dividing the dry weight of the soil by the inner volume of the ring (McKenzie et al. 2002). Soil organic carbon and total nitrogen were determined from air dried 2 g sieved (2 mm) soil. Soil inorganic carbonate content was removed from the soil using 0.5m HCl after which the soil was washed with deionized Ultrapure water for four days. The soil were dried at 40oC for 2 days and then ground in a ball mill to fine powder. The powdered samples were then packed in tin capsules before further analysis in an elemental analyzer (Costech ECS

4010). Soil carbon and nitrogen pools were calculated using soil bulk density per segment depth for amount of soil per hectare multiplied by percent soil organic carbon and nitrogen (Oelbermann and Voroney, 2007). Gross turnover for carbon and nitrogen was determined for nitrogen fertilizer and biobased residues (equation 6.1) at depth 0 -10 cm and 10 - 20 cm.

$$Gross\ turnover\ (years) = \frac{Soil\ organic\ carbon\ or\ nitrogen\ (gm^{-2})}{Annual\ carbon\ or\ nitrogeninputs\ (gm^{-2}year^{-1})} \qquad (6.1)$$

where soil organic carbon or nitrogen is the soil organic pool for carbon or nitrogen in the treatment with nitrogen fertilizer and biobased residues, annual carbon or nitrogen inputs is the measured annual input of organic material from biobased and crop residues.

Residue stabilization efficiency (RSE) in the treatment was determined using equation 6.2

$$RSE\ (\%) = \frac{Soil\ organic\ C\ or\ N_{treatment} - Soil\ organic\ C\ or\ N_{control}}{I_{treatment} - I_{control}} \text{ x } 100 \qquad (6.2)$$

where,

Soil organic C or $N_{treatment}$ = soil organic carbon pool (gm^{-2}) for nitrogen fertilizer and biobased residues, Soil organic C or $N_{control}$ = soil organic carbon or nitrogen pool (gm^{-2}) for no nitrogen fertilizer and biobased residues, $I_{treatment}$ = input (gm^{-2}) of organic material in the nitrogen fertilizer and biobased residues plot (from biobased residues and crop residues) over the entire 3-year period, $I_{control}$ = input (gm^{-2}) of organic material in the no nitrogen fertilizer plot (from crop residues) over the entire 3-year period.

6.2.4 Model performance and statistical analysis

Data from the 3-year field research was used to evaluate model performance. The measured versus simulated values of soil organic carbon and measured versus soil organic carbon stock change was evaluated for model performance (Damasi et al., 2018). The soil organic carbon stock change was based on soil organic carbon simulated values minus soil organic carbon stock at the beginning of the experiment. The soil organic carbon stock change calculation was used to assess and account for the influence of the initialization method adopted for the historical land use (Dimassi et al. 2018). The following statistical test was applied: coefficient of determination (r^2) to measure the strength of the linear relationship between simulated and measured values, root mean square error (RMSE) to estimate the extent of the model error, and Nash-Sutcliffe efficiency (NSE) to measure the agreement between measured and simulated values (range is from ∞ to 1).

Since the goal of this chapter was to evaluate the carbon sequestration potential of biobased residues under continuous cropping and crop rotation, the last 10 years (2158 to 2167) of the simulation was used to quantity differences in active, slow, and passive fractions. This is because the last 10 years represent stabilized carbon where the changes in the fractions are no longer detected by Century (West and Six 2007; Dil and Oelbermann 2014). Parametric statistics was used after the data pass the homogeneity and normality criteria using Shapiro-Wilk test with no violation of assumptions. An analysis of variance was carried out for each site to evaluate if there is any difference between treatments, followed by Tukey pairwise comparison to evaluate the variance in the treatments considering differences to be significant at $p<0.05$. Statistical analysis was done with R v. 4.1.0 (R Core Team, 2020).

6.3 Results

6.3.1 Soil organic carbon stock

The soil organic carbon stock during the initialization period before forest clearing was 13,882 g C m^{-2}. Upon conversion to agriculture in 1915, the soil organic carbon stock increased and was followed by a rapid decline until the year 2017 (Figure 6.1). In 2018, treatments with compost and biosolids had a greater ($p<0.05$) soil organic carbon stock compared to digestate and nitrogen fertilizer (Table 6.2); furthermore, continuous cropping increased soil organic carbon compared to crop rotation ($p<0.05$; Table 6.2). Specifically, continuous cropping with compost had the highest soil organic carbon stock, while crop rotation with nitrogen fertilizer had the lowest soil organic carbon stock ($p<0.05$; Figure 6.1). In 2018, soil organic carbon stock was 11467 g C m^{-2} but ranged from 9348 to 14845 g C m^{-2} in 2167 depending on nitrogen fertilizer or biobased residue treatment (Table 6.2). For example, in 2167, the soil organic carbon stock for compost and biosolids under continuous cropping and crop rotation increased (range; 191 to 3379 g C m^{-2}) except for crop rotation with biosolids (1112 g C m^{-2}) that decreased when compared to year 2018. Soil organic carbon for digestate and nitrogen fertilizer decreased (range; 6 to 2119 g C m^{-2}) in 2167 when compared to year 2018 (Table 6.2). Compared to continuous cropping with nitrogen fertilizer, soil organic carbon increased for continuous cropping with biobased residues and decreased for crop rotation with biobased residues except for that of compost (Table 6.2).

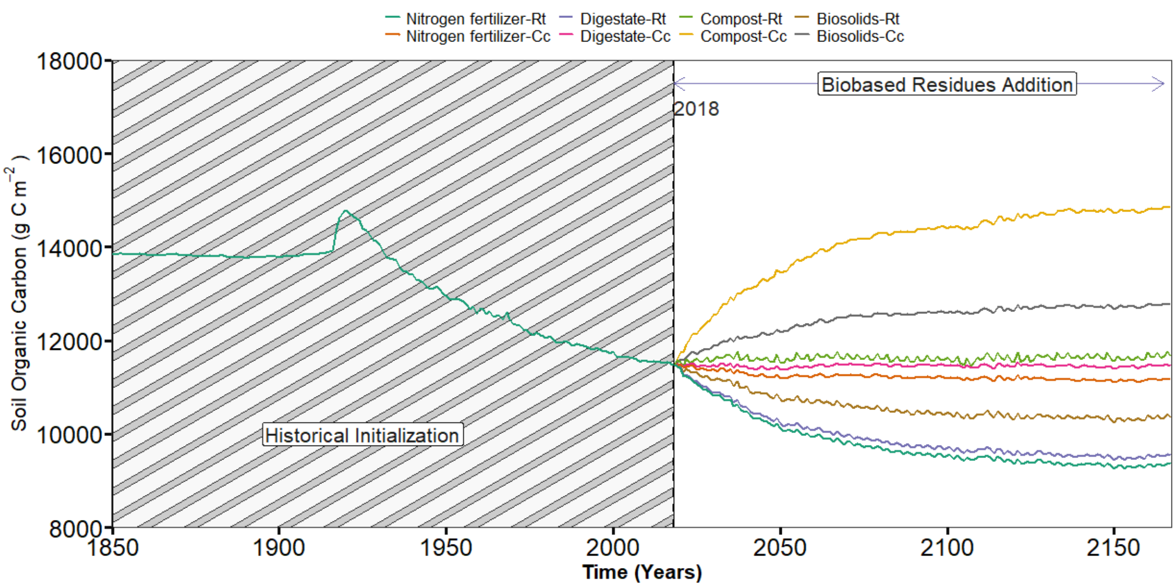


Figure 6.1: Soil Organic Carbon (g C m^{-2}) stocks simulated by Century model showing the historical initialization and the 150 years simulation for the treatments in Elora Research Station, Ontario, Canada. **Note:** Cc, continuous cropping; Rt, crop rotation.

Table 6.2: Changes in soil organic carbon stocks (g m^{-2}) simulated by Century model for the treatments in Elora Research Station, Ontario, Canada.

Treatments	Soil Organic Carbon (g m^{-2})		
	Year 2167	**Change from year 2018**	**Change from Nitrogen Fertilizer_Cc in year 2167**
Nitrogen Fertilizer-Cc	11163[e]	-304	-
Compost-Cc	14845[a]	3379	3683
Biosolids-Cc	12767[b]	1301	1605
Digestate-Cc	11460[c]	-6	298
Nitrogen Fertilizer-Rt	9348[g]	-2119	-1815
Compost-Rt	11658[d]	191	495
Biosolids-Rt	10355[f]	-1112	-808
Digestate-Rt	9534[g]	-1932	-1628

Values with the same letter are not significantly difference (p>0.05) among treatments. **Note:** Cc, continuous cropping; Rt, crop rotation.

6.3.2 Soil organic carbon fractions

During the initialization period, the quantity of carbon in each fraction before conversion to agriculture was 659 (active), 7837 (slow) and 4909 (passive) g C m^{-2} (Figure 6.2). After forest conversion, the active and slow carbon fractions rose briefly and then declined in 2017, while passive fraction increased slightly and remained steady until 2017 (Figure 6.2). Soil organic carbon fractions were greatest in continuous cropping with compost but were lowest in crop rotation with nitrogen fertilizer (p<0.05; Table 6.3). Carbon fractions also varied significantly (p<0.05), depending on amendment type, for active (564 to 1161 g m^{-2}), slow (3703 to 7755 g m^{-2}), and passive (4756 to 5181 g m^{-2}) (Table 6.3). Furthermore, carbon fractions were significantly (p<0.05) greater for compost and biosolids compared to digestate and nitrogen fertilizer. Also, carbon fractions were significantly higher with continuous cropping compared to crop rotation (p<0.05; Table 6.3).

The accumulation of carbon within each fraction varied between the years 2018 and 2167, where the greatest carbon increase in all fractions occurred in continuous cropping with compost and biosolids (p<0.05; Table 6.4). The active carbon fraction showed an increase in carbon, except for crop rotation with nitrogen fertilizer, biosolids and digestate (Table 6.4). The slow carbon fraction increased in continuous cropping with compost and biosolids, and crop rotation with compost. The passive fraction decreases carbon except for continuous cropping with compost and biosolids (Table 6.4).

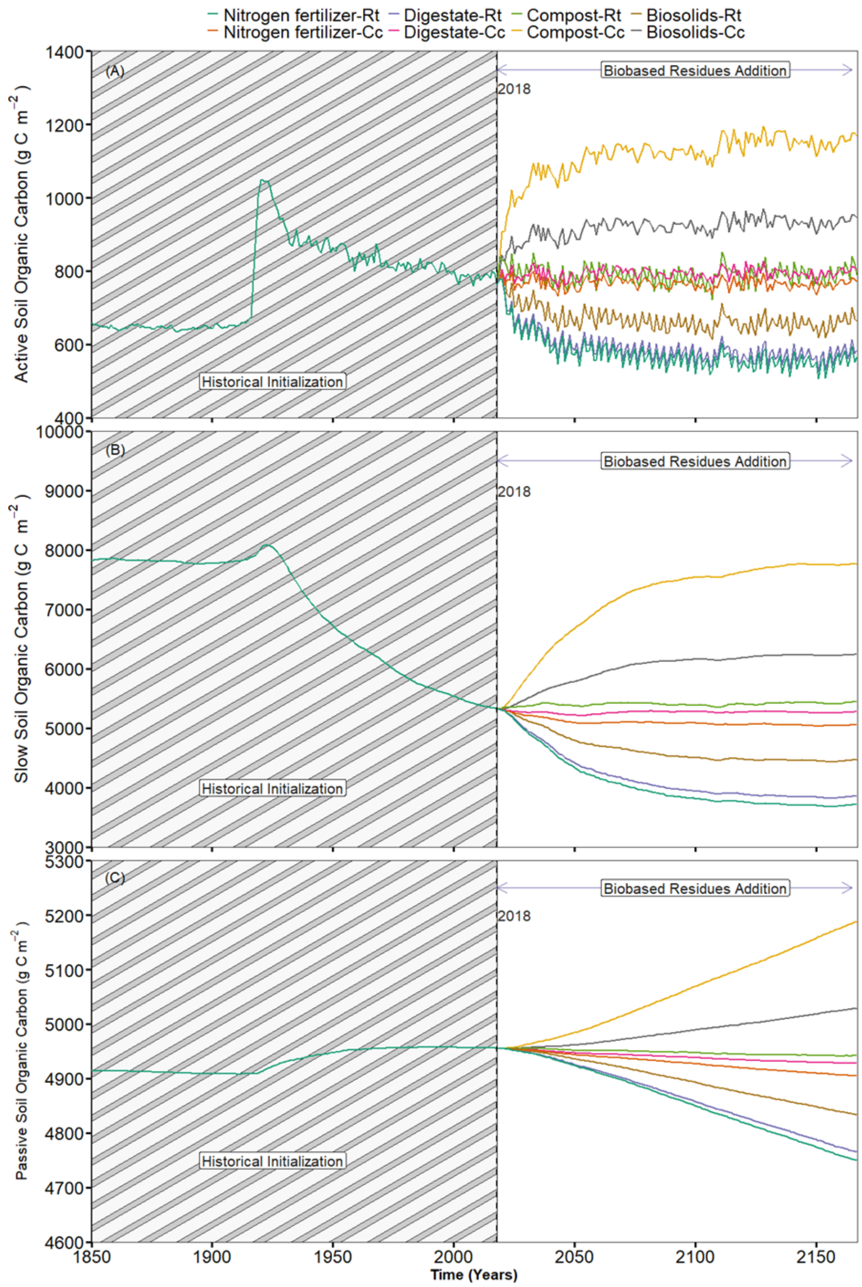


Figure 6.2: Active (a), Slow (b), and Passive (c) SOC fractions simulated by Century model showing the historical initialization and the 150 years simulation for the treatments at Elora Research Station, Ontario, Canada. **Note:** Cc, continuous cropping; Rt, crop rotation.

Table 6.3: Mean values of soil organic carbon (SOC) fractions (g m^{-2}) from year 2158 to 2167 simulated by Century model for the treatments in Elora Research Station, Ontario, Canada. Standard errors are given in parentheses (n=10).

Treatments	Active SOC (g m^{-2})	Slow SOC (g m^{-2})	Passive SOC (g m^{-2})
Nitrogen Fertilizer-Cc	770.40 (2.30)^{d}	5049.6 (2.18)^{e}	4906.0 (0.27)^{e}
Compost-Cc	1160.9 (4.16)^{a}	7754.5 (3.02)^{a}	5180.7 (1.72)^{a}
Biosolids-Cc	941.0 (3.06)^{b}	6232.1 (2.52)^{b}	5026.4 (0.61)^{b}
Digestate-Cc	801.5 (2.44)^{c}	5268.1 (2.22)^{d}	4928.1 (0.11)^{d}
Nitrogen Fertilizer-Rt	563.9 (5.32)^{f}	3703.2 (3.96)^{h}	4755.8 (1.29)^{h}
Compost-Rt	803.1 (7.31)^{c}	5426.7 (4.64)^{c}	4941.7 (0.07)^{c}
Biosolids-Rt	668.4 (5.26)^{e}	4456.5 (4.24)^{f}	4837.3 (0.71)^{f}
Digestate-Rt	583.0 (5.20)^{f}	3841.8 (3.99)^{g}	4770.6 (1.19)^{g}

Values with the same letter are not significantly difference (p>0.05) among treatments. **Note:** Cc, continuous cropping; Rt, crop rotation.

Table 6.4: Mean values of changes in the different soil organic carbon (SOC) fractions (g m^{-2}) between year 2018 and 2167 for the treatments in Elora Research Station, Ontario, Canada.

Treatments	Active SOC (g m^{-2})	Slow SOC (g m^{-2})	Passive SOC (g m^{-2})
Nitrogen Fertilizer-Cc	7.2	-264.5	-50.7
Compost-Cc	404.6	2447.1	233.3
Biosolids-Cc	180.8	920.9	73.7
Digestate-Cc	38.8	-45.6	-27.9
Nitrogen Fertilizer-Rt	-197.9	-1603.0	-205.8
Compost-Rt	24.9	124.0	-13.6
Biosolids-Rt	-100.7	-848.1	-121.6
Digestate-Rt	-180.2	-1464.2	-190.5

Values with the same letter are not significantly difference (p>0.05) among treatments. **Note:** Cc, continuous cropping; Rt, crop rotation.

6.3.3 Model performance, turnover, and residue stabilization

Century overestimated soil organic carbon stocks for compost and biosolids but underestimated it for nitrogen fertilizer and digestate when using the soil organic carbon stock change approach (Figure 6.3). The relationship between soil organic carbon stock change versus measured soil organic carbon was better compared to simulated soil organic carbon versus soil organic carbon (Figure 6.3). The simulated soil organic carbon stocks were 87 to 127 % higher than the measured soil organic carbon stocks (Figure 6.3a). The simulated soil organic carbon stocks overestimated with values ranging from 5328 to 6447 g m^{-2}. The soil organic carbon stock change either under- or over-estimated soil organic carbon stock with values ranging from -26 % to 16 % (Figure 6.3b). In addition, the statistical parameter test (R^2 = 0.99, RMSE = 490.6 g m^{-2}, NSE = -2.81) showed the extent Century model accurately evaluated the soil organic carbon stock at the experiment location (Figure 6.4). The carbon turnover for nitrogen fertilizer was higher and significantly different (p<0.05) from biobased residues except for digestate (Table 6.5). The nitrogen turnover for nitrogen fertilizer was higher and significantly different (p<0.05) from biobased residues (Table 6.5). Carbon residue stabilization was highest for biosolids and lowest for nitrogen fertilizer. Nitrogen residue stabilization was highest for nitrogen fertilizer and lowest for biosolids at depth 0-10 cm, while at depth 10-20 cm, digestate was highest and compost was lowest (Table 6.5).

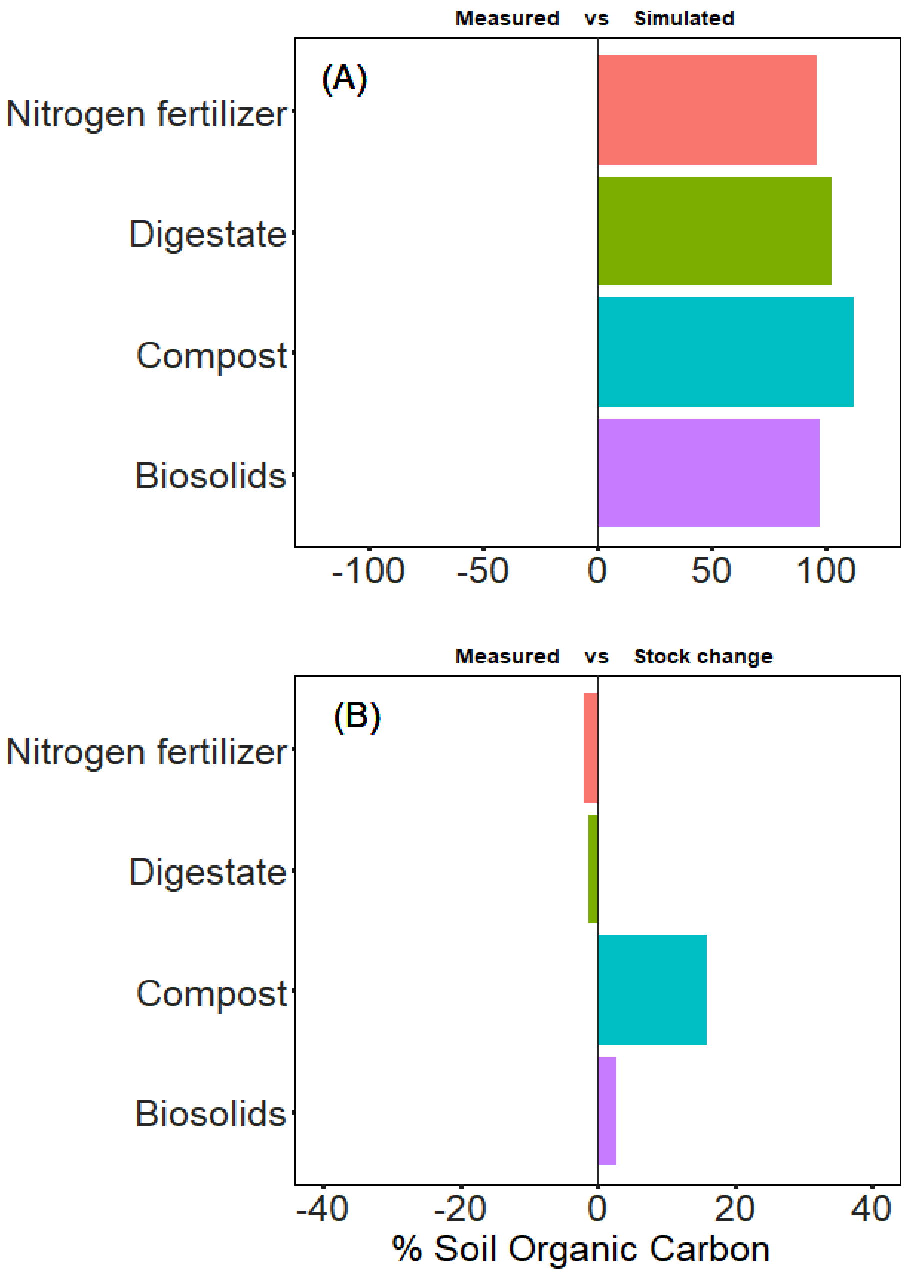


Figure 6.3: Relationship between measured soil organic carbon versus simulated soil organic carbon **(a)** and measured soil organic carbon versus soil organic carbon stock change (**b)** from the 3-yr rotation treatment measurement and simulation in Elora Research Station, Ontario, Canada.

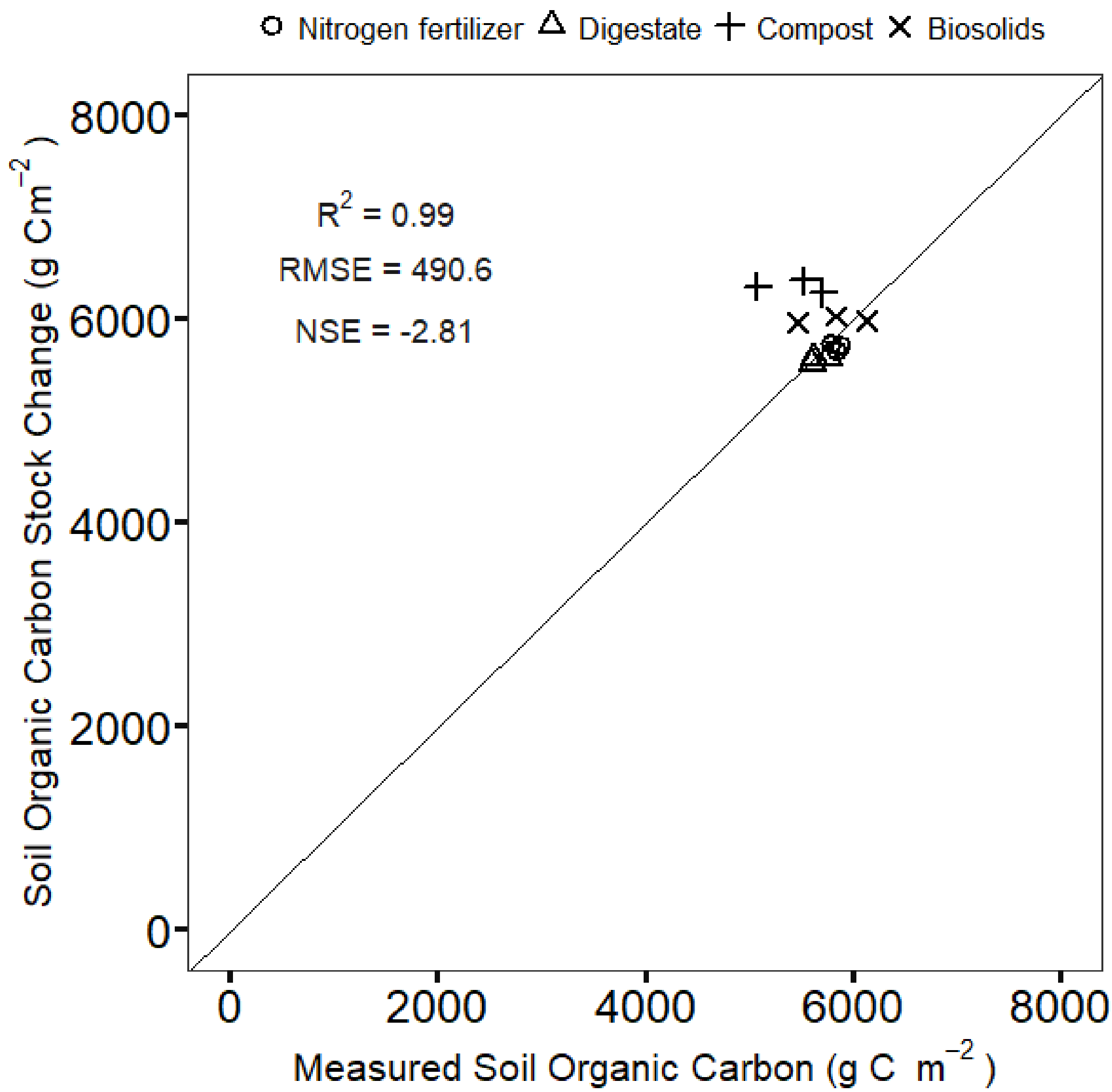


Figure 6.4: Relationship between measured soil organic carbon versus soil organic carbon stock change from the 3-yr rotation measurement and simulation in Elora Research Station, Ontario, Canada.

Table 6.5: Gross soil organic carbon turnover and residue stabilization for nitrogen fertilizer and biobased residues under corn-soybean rotation at Elora Research Station, Ontario, Canada.

	Treatment	Turn over (years)		Residue stabilization (%)	
		0-10 cm	10-20 cm	0-10 cm	10-20 cm
Carbon	Nitrogen Fertilizer	14(1)[a]	12(1)[a]	-1358	-1179
	Digestate	13(2)[a]	13(2)[a]	29	-1748
	Compost	5(0)[c]	4(0)[c]	28	-66
	Biosolids	8(1)[b]	8(0)[b]	66	140
Nitrogen	Nitrogen Fertilizer	16(1)[a]	13(1)[a]	378	76
	Digestate	10(1)[b]	11(1)[a]	82	214
	Compost	10(1)[b]	8(0)[b]	102	-42
	Biosolids	11(1)[b]	11(0)[a]	61	128

6.4 Discussion

6.4.1 Biobased residues and crop rotation impact on SOC stock and its associated fractions

Compost and biosolids-based treatment significantly increased the soil organic carbon stock and its fractions compared to digestate and nitrogen fertilizer. This was likely due to quality and quantity of the carbon input from biobased residues. Compost had the highest carbon input, followed by biosolids and digestate, the lowest (see Table 3.2). Correspondingly, the lower the quality and quantity of biobased residues, the lower the soil organic carbon stock and its fractions and vice versa. However, increasing the application rate of biobased residues has limited climate change mitigation potential and can cause nutrient leaching and pollution (Poulton et al., 2018). Furthermore, the quantity of biobased residues added in this assessment was based on provincial regulations. Similarly, in Europe, Bruni et al. (2021) using Century model across 14 long term agricultural sites ranging from moderately fine to medium texture soils noted that soil organic carbon increase should be based on increase of quality not quantity of exogenous organic matter. This includes compost, biosolids and digestate because of regional scale application limitation, risk of nitrate and phosphate pollution and the large greenhouse gas emissions (Bruni et al., 2021). Therefore, the addition of biobased residues to agricultural land should focus on increasing the quality, not the quantity of soil organic carbon.

The potential of biobased residues was limited with conservation crop rotation compared to conventional continuous cropping. This was likely due to the quality and quantity of crop residue input. Crop residue input from the research site was 1580 g m^{-2} (maize) and 424 g m^{-2} (soybean). These values indicated that under crop rotation, carbon input, despite the additional input of carbon from biobased residues, was lower because soybeans have lower residue input. Crop rotation affected crop residue quality through the inclusion of low carbon/nitrogen ratio crops (e.g., soybean) in the rotation (Kallenbach et al., 2015), which created a favourable environment for microbes with a fast turnover rate and carbon-use efficiency (Zhu et al. 2018). Likewise, in USA, Johnson et al. (2006) found that crop residue input from maize (680 g m^{-2}) was greater compared to soybeans (250 g m^{-2}). Thus, crop rotation regulates the decomposition of the quality of crop residue and biobased residues.

However, crop rotation with compost increased over the 150 years of simulation. The exemption of crop rotation with compost can best be explained by the relatively high carbon/nitrogen ratio of compost when compared to biosolids and digestate. This means carbon/nitrogen ratio of compost slows down decomposition because it is a more recalcitrant sourced carbon compared to biosolids and digestate. McDaniel et al. (2014b) also found that crop rotation regulated carbon and nitrogen dynamics that in turn affected the quality of crop residues, noting that rapid decomposition occurs with the presence of a legume (e.g., soybean). The addition of compost was also found to increase soil organic carbon by 12% over 19 years in a moderately fine and medium texture soil in a maize-tomato (*Lycopersicum esculentum Mill.*) rotation in California. The soil organic carbon increase was attributed to the more stabilized carbon from recalcitrant compounds present in the compost (Tautges et al. 2019).

Using a global database of 67 long-term agricultural experiments, West and Post (2002) found a 19 g $m^{-2}yr^{-1}$ decline in soil organic carbon when converting from continuous cropping to crop rotation under agroecosystems with soil texture ranging from fine to coarse. Álvaro-Fuentes et al. (2009) also found that soil organic carbon loss was greater in a medium texture soil under barley (*Hordeum vulgare L.*) - fallow rotation compared to continuous barley in Spain. However, Schmer et al. (2020) concluded that soil organic carbon (0-150 cm) accretion was significantly greater in a 2-year and 4-year crop rotation compared to continuous cropping after 34 years in a complex moderately fine and medium texture soil in Nebraska, US. They also found that rotation complexity increased soil organic carbon stocks (Schmer et al., 2020). In addition, based on a meta-analysis, McDaniel et al. (2014a) concluded that soil organic carbon stocks were similar between crop rotation compared to continuous cropping under maize production but were greater in continuous cropping for soybean (11%) and sorghum (*Sorghum bicolor*) (8%).

Among the carbon fractions, the allocation of carbon was highest for slow fraction followed by passive and active. Similarly, Dil and Oelbermann (2014) found that slow fraction has the greatest carbon proportion allocation, followed by passive and active under urea ammonium nitrate enriched biochar. The slow and passive fractions increased with biobased residue addition (Vignesh et al., 2012). Oelbermann and Voroney (2011) also observed slow fraction had the highest carbon proportion under temperate and tropical sole crop systems.

6.4.2 Model validation and parameterization

The soil organic carbon stock change improved the performance of Century model compared to the simulated soil organic carbon stocks. This was due to the capacity of soil organic carbon stock change to account for the discrepancies and inconsistencies between measured and simulated values that the century model introduced. Simulated soil organic carbon at the end of the historical initialization period were not shifted to match the measured soil organic carbon at the beginning of 2018 when the evaluation of the treatment began. Similarly, Dimassi et al. (2018) found that using the soil organic carbon stock change instead of absolute simulated soil organic carbon resulted in different Century model performance that was independent of the historical initialization. However, in contrast to this research, Dimassi et al. (2018) found that simulated versus measured soil organic carbon either under- or over-estimated the soil organic carbon stocks (-42 to 33 %) compared to only overestimated soil organic carbon observed in this chapter's result (87 to 127 %). The authors also observed that soil organic carbon stock change versus measured consistently underestimated soil organic carbon stock (-142 to - 67 %) compared to either under- or over-estimated soil organic carbon stock (-26 % to 16 %) in this investigation.

They further noted a poor fit linear relationship of soil organic carbon stock change versus measured soil organic carbon (R^2 = 0.17) compared to strongly fit relationship observed in this evaluation (R^2 = 0.98). These differences can best be explained by the accuracy of the information used to initialize the historical land-use and land management practices. For example, some of the historical information was broadly associated with the region where the experiment was located are limited, not site-specific and detailed like that of Dimassi et al. (2018). Hence, making the soil organic carbon stock change versus measured soil organic carbon validate the model performance regardless of the historical initialization and better suited for this assessment compared to simulated versus measured soil organic carbon.

The carbon and nitrogen turnover for nitrogen fertilizer was notably different from that of the biobased residues except for carbon turnover of digestate. This was likely due to the varied nutrient availability and the carbon-nitrogen ratio of nitrogen fertilizer and biobased residues that influences their microbial carbon use efficiency. Although the application rate of nitrogen fertilizer was lowest compared to biobased residues, the nutrient availability was highest for nitrogen fertilizer and lowest for compost, while the carbon-nitrogen ratio was lowest for nitrogen fertilizer and highest for compost (See Table 3.1, 3.2). This indicated that nitrogen fertilizer had the highest microbial carbon use efficiency and compost had the lowest. Manzoni et al. (2012) observed that substrates with high carbon-nitrogen ratios or nitrogen limiting circumstances result in low microbial carbon use efficiency. Spohn et al (2016) also observed that nitrogen fertilized treatment addition led to increase in microbial carbon use efficiency. Therefore, microbial carbon use efficiency induced by nitrogen fertilizer and biobased residues affect soil organic carbon turnover and long-term soil organic carbon dynamics (Wang and Luo, 2021).

6.4.3 Uncertainty, limitation and implication

The overestimation of simulated soil organic carbon in this assessment confirmed that the Century model has limitations and uncertainties. For instance, accurate prediction of the soil organic carbon and its rate of change was influenced by the pure theoretical determination of the soil organic carbon fractions, instead of direct measurements (Zimmermann et al. 2007; Foereid et al. 2012). The difference in the size of the soil organic carbon fractions also caused identifiability problems where the multiple combinations of the estimated fractions generate similar probability distributions for the measured variable used in the model initialization (Tang and Zhuang 2008; Sierra et al., 2015). In addition, the initialized soil organic carbon and its fractions using equilibrium initialization to approximate the steady-state instead of direct measurement through soil organic carbon fractionation experiment introduced disturbance and the long turnover rates for the recalcitrant components of soil organic carbon and its fractions (Wutzler and Reichstein 2007, 2008). Further research should explore various calibration methods and approaches to optimize model input such as size of the soil organic carbon fractions.

This chapter focused on the long-term (150 years) simulation of carbon dynamics in continuous and crop rotation systems with biobased residues addition with only 3 years data for validation. However, the three years were not enough to capture the reproducibility of the measured by the simulated even though the model was relevant. This may explain why the Nash-Sutcliffe efficiency (NSE) was negative (Figure 6.4). The simulated soil organic carbon stocks at the end of the initialization did not match the soil organic carbon at the beginning of the experiment in 2018. This work addressed the challenge by using the change in the soil organic carbon stock to avoid the mismatch between the simulated and the measured soil organic carbon (Dimassi et al., 2018). Therefore, the simulation of cropping systems with biobased residues addition using Century model addressed in this chapter is a potential pathway to defining science-based policies for carbon neutral acts or markets based on the conceptual model of carbon sequestration.

6.5 Conclusions

The chapter evaluated how three different types of biobased residues would impact soil organic carbon and its associated fractions under continuous cropping and crop rotation on silt loam in southern Canada. The crop rotation treatments showed the greatest limitation in the long-term accretion and stabilization of soil organic carbon but can be improved if biobased residues with a higher carbon/nitrogen ratio and organic matter content are added. The compost and biosolids increased soil organic carbon stock, and the anaerobic digestate resulted in soil organic carbon loss. Also, the carbon accumulation was similar in slow and passive fractions. Using soil organic carbon stock change improved the century model performance compared to the simulated soil organic carbon under limited site-specific historical data and the 3 years of the experiment. Further research should consider broader and longer (> 5 years) evaluation of the contribution of biobased residues and the cropping systems at national and global scales using sites with detailed historical information while accounting for emissions due to biobased residues to achieve carbon sequestration.

Chapter 7
Conclusions and Future Recommendations

7.1 Major Research Findings

This dissertation explores the capacity of biobased residues in relation to soil health function and greenhouse gas emissions under a changing climate – with the intention of providing alternative fertilizer sources for farmers to address soil health and reduce nitrogen fertilizer use. Using a research site with silt loam texture, I found that biobased residues could serve as an alternative fertilizer source for farmers with the potential to improve soil carbon sequestration that reduces soil degradation.

Chapter 2 reviewed biobased residues in relation to soil security. The chapter established the links and assessment factors between sustainable indicators and biobased residues for soil security. This chapter identified that biobased residues have the capacity to address the underlying issues related to humans, the waste they produce, and their application to agricultural soil to ensure food security, although the understanding of biobased residues and their interaction in agricultural is limited. This review reveal biobased residues approach could sufficiently address soil security.

Chapter 3 evaluated the annual greenhouse gas emissions from biobased residues' amended soil in the temperate region. Greenhouse gas sampling and soil ancillary measurement from a field test of biobased residues (compost, biosolids, digestate) and nitrogen fertilizer (urea) were used. The findings of this chapter demonstrate that biobased residues have a lower CH_4 and N_2O emission compared to nitrogen fertilizer during the non-growing season. Non-growing season significantly influences annual greenhouse gas emissions by about 19% to 91%. Soil variables (temperature, moisture, electrical conductivity, nitrate, and ammonium) are sufficient predictive factors for N_2O and CO_2 but not CH_4. Emission depends on all-year-round variable supply of carbon and nitrogen substrate from the nitrogen fertilizer and biobased residues. The freeze-thaw conditions significantly influence emissions during the non-growing season.

Chapter 4 evaluated the effects of spring freeze-thaw events on greenhouse gas emissions from soil amended with biobased residues. The findings of this chapter showed dry phase during freeze-thaw caused intensified CO_2 fluxes compared to the wet and waterlogged freeze-thaw phase due to enhanced warming. Similar to Chapter 3, soil amended with biobased residues may either increase or reduce greenhouse gas fluxes during spring freeze-thaw events depending on the organic material source and production method. Mineralized nutrients for emission are less available in biobased residues compared to nitrogen fertilizer.

Chapter 5 determined the impact of biobased residues on soil health. This was assessed through a conceptual assessment approach and field sites in Ontario and Quebec. Crop productivity was not different between the biobased residues and nitrogen fertilizer. Microbial utilization: biosolids > digestate > compost ≈ nitrogen fertilizer. Soil health score established that biosolids, digestate and nitrogen fertilizer contribute to one or more components of crop productivity.

Chapter 6 determined the capacity of biobased residues to contribute to the soil organic carbon in temperate soil. Century Soil Organic Matter Model was used to assess eight scenarios that incorporated biobased residues and nitrogen fertilizer. The projection over 150 years showed that scenarios with compost and biosolids improved the long-term stabilization of soil organic carbon.

7.2 Key Research Contributions

Apart from the significant and original contribution to the literature, this research aimed to contribute to the farm-level measurement through quantitative evaluation that helped address different sustainable development indicators such as soil degradation and greenhouse gas emission. The chapters provided knowledge on how the different products derived from organic materials diverted from landfills can improve soil health and their role in mitigating GHG emissions while improving soil health, under the current environmental regime and future climatic conditions. The priority of this research was to re-enact and emphasize renewed interest in recycling nutrients from municipal, landfill, industrial and agricultural wastes for soil function and nitrogen fertilizer alternatives.

Each chapter presented in this dissertation was framed to address related agricultural productivity and sustainability challenges. One of the main goals was to provide the biophysical knowledge that will contribute to the development of best management practices that maximize soil health and support agricultural sustainability, increasing resilience to climate change. This is why the studies focused on providing knowledge on how the different products, derived from organic materials diverted from landfills, can improve soil health under the current environmental regime and future climatic conditions. At the same time, assessing their role in mitigating greenhouse gas emissions.

The projects within this dissertation focused mainly on quantitative assessment for a holistic perspective since most efforts on biobased residues, and their carbon sequestration capacity is based on qualitative assessment (Murphy, 2015, Minasny et al., 2017). Apart from the literature review, this dissertation focused mainly on addressing greenhouse gas emissions (annual, spring freeze-thaw), soil health, and prediction of soil organic carbon under sustained climate future scenarios of 150 years.

The studies addressed the challenge of negative reports and sufficiency of biobased residues to help mitigate greenhouse gas emissions (Chapters 3 and 4), improve soil health and crop productivity (Chapter 5), and sequester and stabilize soil organic carbon over the long term (Chapter 6). Also, the residues application rates are feasible and practicable for application in agricultural land in Ontario due to local availability and proximity, cheap cost of transportation and no metal contamination in soil.

Overall, biobased residues can serve as alternatives to conventional mineral fertilizer considering the soil health, crop productivity and greenhouse gas tradeoffs. Agricultural soil with biobased residues can reduce annual greenhouse gas emissions. Non-growing season significantly influences annual greenhouse gas emissions by about 19% to 91%. Biobased residues sustain crop productivity and soil health in a maize-soybean rotation. Compost and biosolids increase long-term soil organic carbon stocks.

7.3 Future Research

The qualitative evaluation of the use of biobased residues provided further insight into understanding the contribution of background nitrogen-sources to greenhouse gas emissions especially during non-growing season, while also contributing to future governance and policy development and agronomic productivity. Future research should include isotopic labelling to trace the contribution of the nitrogen from the crop residues and organic nitrogen to greenhouse gas emissions. Applying machine learning approach for soil health assessment such as feature selection and estimation set to determine the best soil indicators under different management practices and soil around the world. Classification of the impact of biobased residues on microbial community structure and diversity should be expanded using genetic sequencing and lipid analysis. Broader and longer soil carbon sequestration research of biobased residues and cropping systems at national and global scale.

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Appendices

Supplementary data

Chapter 3 Supplementary data

Table S3.1: Nitrogen (N) applied calculation using biobased residues analysis from the first year of the research. Name for data on file; compost (AIMCalgary.pdf), Biosolids (2018 Southgate.pdf), Digestate (Bio-En.jpg).

Biobased residues	Nitrogen (N) applied calculation
Compost	12 t/ha * 2.0% N by weight
	= 12 000 kg/ha * 0.02
	= 240kg/ha
Biosolids	28000L /ha * 1.09 kg/L (density) * 14.2% dry matter content
	* 4.96% N by dry weight
	= 28000L /ha * 1.09 kg/L * 0.142 * 0.0496
	= 215 kg/ha
Digestate	42000L/ha * 0.264 gallons/L / 2.47 acre/ha / 1000 gallons
	* 51.56 kgN/ha
	= 231 kg/ha

Table S3.2: P-value for analysis of variance for nitrous oxide - N_2O, carbon dioxide - CO_2, methane - CH_4 and soil parameters (soil moisture - SM, soil temperature - ST, electrical conductivity - EC, soil ammonium - NH_4^+-N and nitrate - NO_3^--N) at Elora Research Station, Ontario, Canada. p<0.05 are bolded.

Source of variation	**N_2O**	**CO_2**	**CH_4**	**SM**	**ST**	**EC**	**NH_4^+-N**	**NO_3^--N**
Year	0.246	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.021**	**<0.001**	**<0.001**
Season	0.773	**<0.001**	0.144	**<0.001**	**<0.001**	**<0.001**	0.265	**<0.001**
Treatment	0.803	0.996	0.865	0.969	0.999	0.073	**<0.001**	**0.001**
Season x Treatment	0.080	0.687	0.746	0.898	0.997	0.789	0.560	0.520
Year x Treatment	0.451	0.863	0.273	0.993	1.000	**0.006**	**<0.001**	**<0.001**
Year x Season	0.640	0.154	0.778	-	-	-	**<0.001**	**0.002**
Year x Season x Treatment	0.800	0.733	0.610	-	-	-	**0.011**	0.256

Chapter 4 Supplementary data

Table S4.1: The soil greenhouse gas fluxes sampling dates with weather event and visual observation during the spring freeze-thaw event at Elora Research Station, Ontario, Canada.

Spring Freeze thaw Phase	Sampling date	Weather event and visual observation	Visual image
Waterlogged (SFT1)	Mar. 11	The first day the field was accessible for sampling. Rainfall (about 1cm) overnight further melt the snow cover to about 0.5 cm	
	Mar. 12	snow-free but waterlogged soil. Ambient temperature >10^{o}C	
	Mar. 13	Waterlogged with ice flakes. Ambient temperature <0^{o}C	
	Mar. 14	Waterlogged with ice flakes. Ambient temperature <0^{o}C	
	Mar. 15	Waterlogged with ice flakes. Ambient temperature <0^{o}C	
Wet (SFT2)	Mar. 16	Wet with ice flakes. Ambient temperature <0^{o}C	
	Mar. 17	Wet with ice flakes, Ambient temperature about 5^{o}C	

	Mar. 18	Wet with dry patches and no Ice flakes Ambient temperature dropped to 0^{o}C	
	Mar. 19	Wet with dry patches and no Ice flakes. Ambient temperature around 0^{o}C	
	Mar. 20	Wet with dry patches and no Ice flakes. Ambient temperature around 2^{o}C	
	Mar. 21	Wet with dry patches and no Ice flakes. Ambient temperature about 6^{o}C	
Dry (SFT3)	Mar. 22	Dry with wet patches and no Ice flakes. Ambient temperature about 8^{o}C	
	Mar. 23	Dry with wet patches and no Ice flakes. Ambient temperature about 10^{o}C	
	Mar. 24	Dry with wet patches and no Ice flakes Ambient temperature about 13^{o}C	

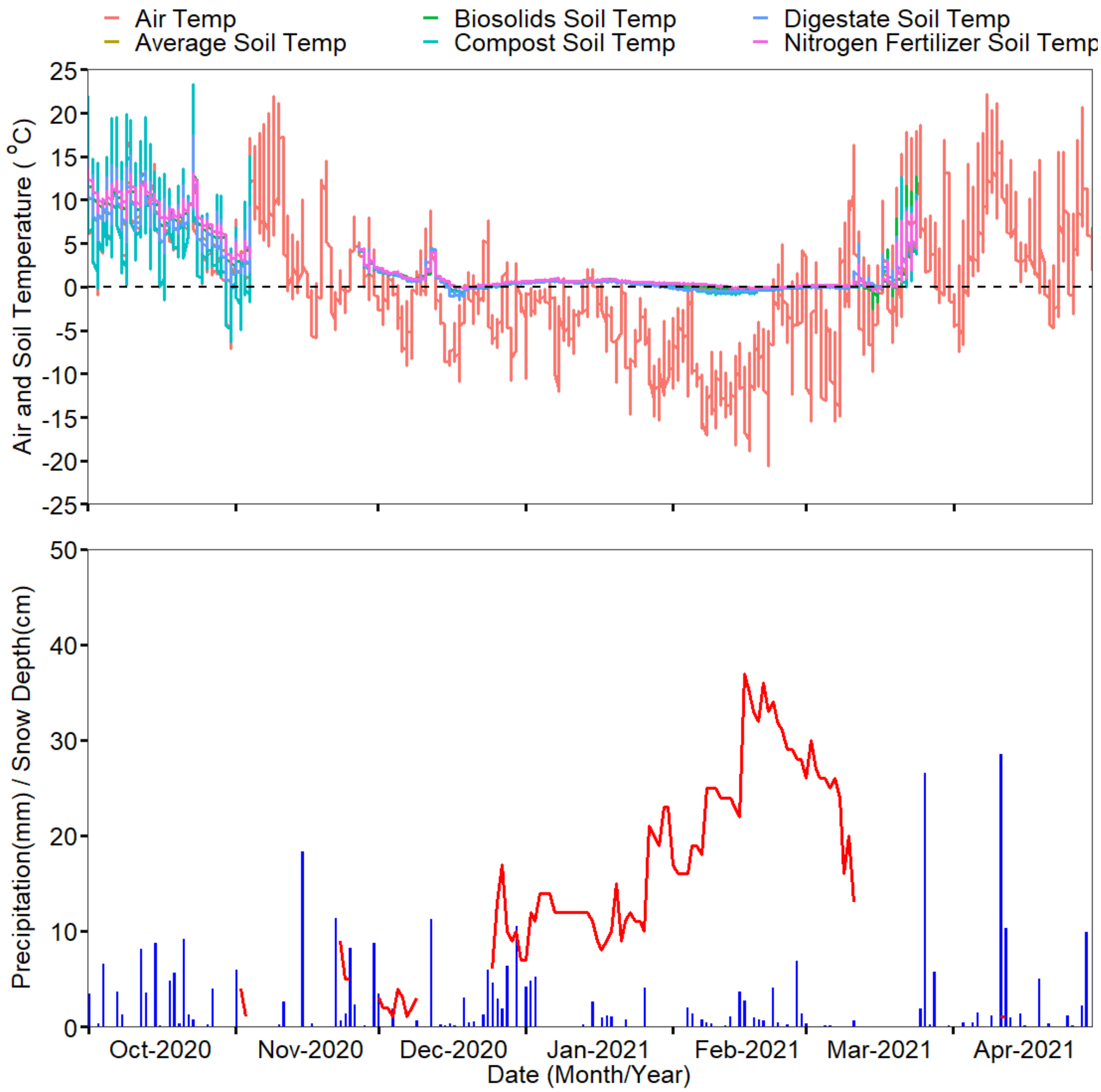

Figure S4.1. Meteorological conditions (a) soil temperature from the four treatment plots (~10cm) and air temperature; and (b) daily precipitation (blue bar) and snow cover depth (red) during the experiment period (from 1 October 2020 to 30 April 2021) at Elora Research Station, Ontario, Canada.

Chapter 5 Supplementary data

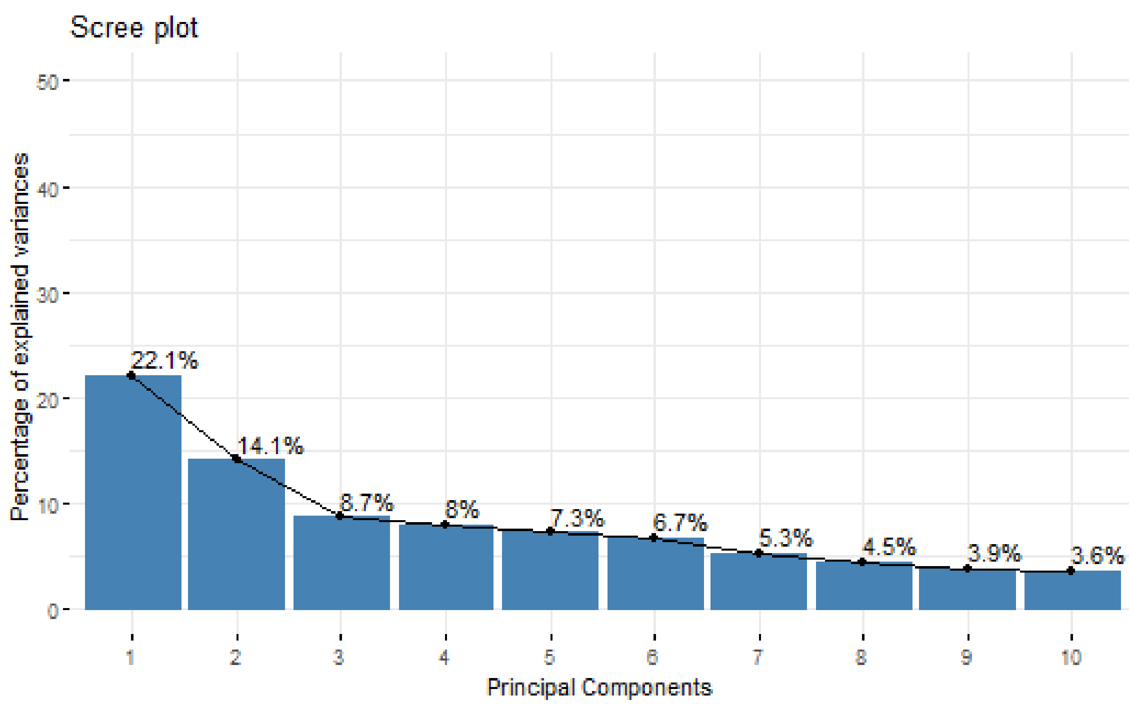

Figure S5.1: Scree plot of eigenvalues against the principal components for soil indicators under nitrogen fertilizer-biobased residues application at Elora Research Station, Ontario, Canada.

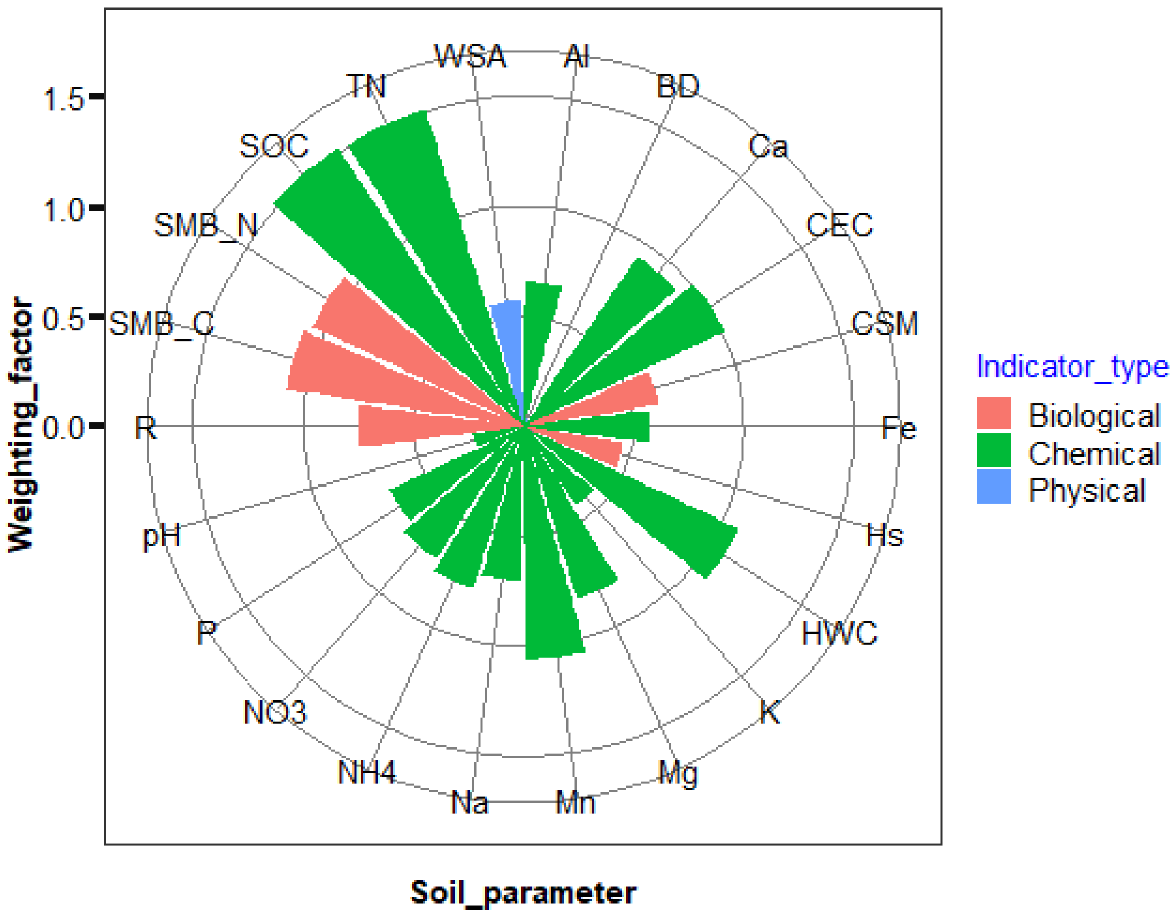

Figure S5.2: Weighting factors for each soil indicator used to compute the Soil Health Assessment at Elora Research Station, Ontario, Canada.

Table S5.1: Analysis of variance of the effects of Treatment, Year, Depth, and their interactions on soil indicators at Elora Research Station, Ontario, Canada.

Soil indicators	Treatment	Year	Depth	Treatment x Year	Treatment x Depth	Year x Depth	Treatment x Year x Depth
Bulk density	0.9	**12.8*** **	1.0	2.1	0.2	5.0**	**3.5** **
WSA (%)	**4.7** **	**280.3*** **	**32.0*** **	0.4	**6.0** **	**107.9*** **	**3.8** **
Porosity (%)	0.9	**12.6*** **	1.1	2.1	**0.2**	**5.1** **	**3.5** **
Infiltration rate (mm hr^{-1})	1.4	0.3	-	0.2	-	-	-
pH	0.1	**48.0*** **	2.1	0.6	**3.7* **	1.3	0.6
NO_3—N (mg kg^{-1})	**5.4** **	**33.9*** **	**5.1* **	**3.9** **	0.6	2.9	1.1
NH_4^+-N(mg kg^{-1})	2.3	**167.6*** **	**24.0*** **	0.9	**4.5** **	**7.4** **	1.3
PO_4^{3-}-P	**2.9* **	**17.7*** **	**59.1*** **	1.5	**4.0* **	0.3	1.0
SOC	2.2	**12.2*** **	0.8	0.3	0.1	0.3	0.1
TN	1.5	**10.7*** **	1.3	0.4	0.1	0.7	0.2
CEC	1.7	**98.3*** **	**7.7** **	**2.4* **	1.0	**12.3*** **	1.2
Ca	1.3	**133.7*** **	2.5	1.2	1.8	1.9	0.5
Mg	**4.5** **	**10.1*** **	0.0	0.6	0.2	**4.4* **	0.5
K	**4.0* **	**35.3*** **	**21.2*** **	**4.9*** **	**5.4** **	**39.7*** **	**4.2** **
Na	**7.5*** **	**87.8*** **	**16.0*** **	**7.5*** **	1.1	**16.0*** **	1.1
Al	1.2	**43.2*** **	0.0	1.2	2.4	0.0	**2.4* **
Mn	1.5	2.4	2.4	1.5	1.5	2.4	1.5
Fe	1.3	**35.3*** **	0.0	1.3	**5.4** **	0.0	**5.4*** **
HWC	**3.0* **	**64.1*** **	**67.2*** **	0.9	2.6	2.1	1.9
SMB-C	0.6	**5.8** **	**15.1*** **	1.4	0.6	1.7	1.3
SMB-N	1.2	0.4	0.3	**2.4* **	0.7	2.6	1.2

The values shown are F-values. The F values in bold text indicate that effects were significant at the 0.05 level or lower. ns: not significant. *, **, *** indicate p<0.05, p<0.01 and p < 0.001, respectively. **Note:** WSA, water stable aggregates; SOC, soil organic carbon; TN, total nitrogen; CEC, cation exchange capacity; HWC, hot water extractable carbon; SMB-C, soil microbial biomass carbon; SMB-N, soil microbial biomass nitrogen.

Table. S5.2: Soil indicators that were directly affected by nitrogen fertilizer and biobased residues at depth 0-10 cm in Elora Research Station, Ontario, Canada.

Treatment	Year	WSA (%)	NO_3^--N (mg kg^{-1})	PO_4^{3-}—P (mg kg^{-1})	Mg (cmol kg^{-1})	K (cmol kg^{-1})	Na (cmol kg^{-1})	HWC (mg kg^{-1})
Nitrogen Fertilizer	Year1	**44.3 (3.5)^{a}**	17.7 (4.5)	20.6 (6.7)	4.5 (0.1)	0.9 (0.4)	**0.1 (0.0)^{b}**	**381.0 (36.4)^{a}**
	Year2	31.7 (7.0)	8.0 (0.6)	19.0 (4.0)	3.7 (0.1)	0.5 (0.1)	0.0 (0.0)	305.6 (13.2)
	Year3	15.7 (4.5)	4.7 (0.2)	**6.2 (1.1)^{b}**	4.1 (0.2)	0.6 (0.1)	0.0 (0.0)	190.8 (23.1)
	Year (1-3)	**30.6 (4.4)^{a}**	10.1 (2.2)	15.3 (3.1)	4.1 (0.1)	0.7 (0.1)	0.0 (0.0)	292.5 (27.2)
Digestate	Year1	**21.8 (3.8)^{b}**	16.1 (3.8)	9.3 (1.1)	3.9 (0.4)	0.5 (0.2)	**0.2 (0.0)^{ab}**	**237.6 (35.5)^{b}**
	Year2	17.3 (5.2)	7.4 (0.6)	11.4 (1.2)	3.2 (0.1)	0.4 (0.0)	0.0 (0.0)	292.7 (10.7)
	Year3	9.4 (1.0)	9.1 (3.4)	**5.0 (0.9)^{b}**	3.7 (0.4)	0.6 (0.1)	0.0 (0.0)	167.2 (13.8)
	Year (1-3)	**16.1 (2.5)^{b}**	10.9 (1.9)	8.6 (1.0)	3.6 (0.2)	0.5 (0.1)	0.1 (0.0)	232.5 (19.5)
Compost	Year1	**35.9 (4.2)^{ab}**	11.4 (3.7)	9.3 (0.9)	3.9 (0.1)	1.2 (0.8)	**0.3 (0.1)^{a}**	**332.0 (34.7)^{ab}**
	Year2	26.8 (3.6)	6.6 (0.4)	13.0 (0.4)	3.5 (0.3)	0.5 (0.0)	0.0 (0.0)	285.2 (19.7)
	Year3	10.6 (1.0)	4.7 (0.7)	**3.8 (0.4)^{b}**	3.4 (0.2)	0.5 (0.0)	0.0 (0.0)	190.3 (15.7)
	Year (1-3)	**24.4 (3.6)^{ab}**	7.6 (1.4)	8.7 (1.2)	3.6 (0.1)	0.7 (0.3)	0.1 (0.0)	269.2 (22.0)
Biosolids	Year1	**22.9 (2.8)^{b}**	13.4 (3.4)	12.8 (2.1)	4.1 (0.2)	0.8 (0.3)	**0.3 (0.1)^{ab}**	**349.7 (25.8)^{ab}**
	Year2	30.5 (2.3)	6.5 (0.6)	20.5 (4.3)	3.6 (0.2)	0.4 (0.0)	0.0 (0.0)	305.1 (29.6)
	Year3	13.5 (1.7)	4.4 (0.6)	**14.0 (3.3)^{a}**	4.2 (0.2)	0.6 (0.1)	0.0 (0.0)	202.7 (18.8)
	Year (1-3)	**22.3 (2.4)^{ab}**	8.1 (1.6)	15.8 (2.0)	4.0 (0.1)	0.6 (0.1)	0.1 (0.1)	285.8 (22.7)

Values are means with standard errors in bracket. Mean with significantly difference (p<0.05) among treatments within year are bolded and indicated with different letters. **Note:** WSA, water stable aggregates; NO_3^--N, nitrate; PO_4^{3-}-P, orthophosphate; Mg, magnesium, K, potassium; Na, sodium; HWC, hot water extractable carbon.

Table. S5.3: Soil indicators that were directly affected by nitrogen fertilizer and biobased residues at depth 10-20 cm in Elora Research Station, Ontario, Canada.

Treatment	Year	WSA (%)	NO_3^--N (mg kg^{-1})	PO_4^{3-}-P (mg kg^{-1})	Mg (cmol kg^{-1})	K (cmol kg^{-1})	Na (cmol kg^{-1})	HWC (mg kg^{-1})
Nitrogen Fertilizer	Year1	59.7 (4.4)	**26.4 (3.7)**[a]	4.1 (1.1)	4.1 (0.2)	0.3 (0.1)	**0.0 (0.0)**[b]	250.1 (43.0)
	Year2	17.6 (2.7)	6.3 (0.4)	6.7 (2.6)	3.5 (0.3)	**0.4 (0.1)**[b]	0.0 (0.0)	225.2 (14.3)
	Year3	14.3 (4.0)	**8.6 (1.9)**[ab]	-0.8 (0.3)	4.4 (0.2)	**0.4 (0.0)**[b]	**0.0 (0.0)**[b]	**162.7 (10.9)**[a]
	Year (1-3)	30.6 (6.5)	13.8 (3.0)	3.3 (1.3)	4.0 (0.2)	0.4 (0.0)	0.0 (0.0)	212.7 (17.9)
Digestate	Year1	68.1 (1.9)	**18.8 (2.6)**[ab]	3.8 (0.3)	3.0 (0.8)	0.3 (0.1)	**0.1 (0.0)**[ab]	252.2 (21.0)
	Year2	16.2 (0.9)	6.0 (0.4)	3.2 (0.6)	3.4 (0.3)	**0.6 (0.0)**[a]	0.0 (0.0)	209.1 (15.3)
	Year3	9.8 (1.8)	**15.4 (1.5)**[a]	-0.1 (0.4)	4.2 (0.4)	**5.6 (0.6)**[a]	**0.0 (0.0)**[ab]	**144.0 (8.4)**[ab]
	Year (1-3)	31.4 (7.9)	13.4 (1.9)	2.3 (0.6)	3.5 (0.3)	2.1 (0.8)	0.0 (0.0)	201.7 (15.7)
Compost	Year1	63.3 (3.1)	**8.8 (2.5)**[b]	2.3 (0.5)	3.7 (0.1)	0.3 (0.0)	**0.1 (0.0)**[ab]	214.2 (9.6)
	Year2	16.2 (0.8)	7.2 (0.5)	13.5 (8.1)	3.3 (0.2)	**0.3 (0.0)**[b]	0.0 (0.0)	229.1 (14.1)
	Year3	10.6 (1.4)	**7.1 (0.5)**[b]	-0.1 (0.3)	4.2 (0.1)	**3.5 (1.9)**[ab]	**0.0 (0.0)**[ab]	**105.2 (6.5)**[b]
	Year (1-3)	30.0 (7.2)	7.7 (0.8)	5.3 (3.0)	3.7 (0.1)	1.4 (0.7)	0.0 (0.0)	182.9 (17.6)
Biosolids	Year1	68.1 (2.1)	**12.9 (4.0)**[ab]	3.3 (0.9)	4.0 (0.2)	0.3 (0.0)	**0.1 (0.0)**[a]	228.5 (19.0)
	Year2	13.2 (1.6)	6.4 (0.4)	5.6 (2.2)	3.4 (0.2)	**0.4 (0.0)**[b]	0.0 (0.0)	197.5 (11.5)
	Year3	11.1 (1.1)	**11.5 (2.2)**[ab]	0.0 (0.4)	4.5 (0.2)	**5.1 (0.1)**[a]	0.0 (0.0)[a]	**148.6 (16.5)**[ab]
	Year (1-3)	30.8 (8.0)	10.3 (1.6)	3.0 (1.0)	4.0 (0.2)	1.9 (0.7)	0.0 (0.0)	191.5 (13.0)

Values are means with standard errors in bracket. Mean with significantly difference ($p<0.05$) among treatments within year are bolded and indicated with different letters. **Note:** WSA, water stable aggregates; NO_3^--N, nitrate; PO_4^{3-}-P, orthophosphate; Mg, magnesium, K, potassium; Na, sodium; HWC, hot water extractable carbon.

Chapter 6 Supplementary data

Table S6.1: Soil characteristics (0-20cm) before the start of the experiment at Elora Research Station, Ontario, Canada.

Soil Parameters	Values
Sand (%)	21
Silt (%)	55
Clay (%)	24
Bulk density (g cm^{-3})	1.1
pH	7.9
Soil organic carbon (%)	2.5
Soil total N (%)	0.2
C/N	10
Soil Ammonium, NH_4^+ (mg kg^{-1})	6.5
Soil nitrate, NO_3^- (mg kg^{-1})	11.7
Soil microbial biomass, SMB-C (µg C g^{-1})	870
Soil microbial biomass, SMB-N (µg C g^{-1})	96.7

Table S6.2: Climate data adopted for the 150-year Century simulation at Elora Research Station, Ontario, Canada based on a 17-year average.

Month	Temperature (°C)		Precipitation (cm)
	Min	Max	
1	-10.3	-3.0	7.5
2	-10.6	-2.6	6.0
3	-5.6	3.1	6.7
4	0.9	10.9	8.1
5	7.1	18.4	7.5
6	12.2	23.2	7.2
7	14.3	25.8	7.4
8	12.9	24.4	6.9
9	9.6	21.1	7.3
10	4.0	13.6	8.7
11	-1.2	6.2	6.9
12	-5.9	-0.2	6.4