Land Application of Municipal Wastewater Biosolids in Canada: A Carbon Footprint Assessment

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A thesis submitted to McGill University in partial fulfillment of the requirements of the degree of Master of Science.

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Abstract

As more Canadian jurisdictions ban the landfilling of organic material, there is increasing pressure to recycle municipal wastewater biosolids to agricultural soils to replace commercial fertilizers. However, information gaps remain about the climate change impact of this practice. A carbon footprint analysis was conducted to quantify the climate change impact associated with the processing and land application of three different types of biosolids: digested, composted, and alkaline biosolids. The biosolids were applied to agricultural land on McGill's Macdonald Campus Research Farm near Montreal, Canada. OpenLCA 1.11 coupled with the life cycle inventory database Ecoinvent 3.6 were used to perform a carbon footprint assessment of scenarios including each of the three processing methods to determine their global warming impact. The comparative analysis revealed different results depending on the default disposal scenario (i.e., depending on which avoided emissions were considered in the analysis). In the first case, there was no consideration of avoided emissions from sludge disposal. In this case, the scenario with the least climate change impact was the application of urea fertilizer (positive control), followed closely by digested biosolids. In the second case, if the avoidance of emissions from the incineration of sludge was considered, the treatment scenario with the least climate change impact was application of digested biosolids. Finally, if the avoidance of emissions from landfilling of sludge was considered, the scenario with the least climate change impact was the application of composted biosolids. The results highlight the importance of diverting organic material from landfill or incineration, as well as the greenhouse gas emissions potentially associated with or avoided through the treatment and land application of sewage sludge. All in all, a holistic assessment including upstream processes and avoided emissions is essential for a fair comparison of different treatment and application scenarios.

Résumé

Étant donné que de plus en plus de juridictions canadiennes interdisent la mise en décharge de matières organiques, la pression augmente pour recycler les biosolides des eaux usées municipales dans les sols agricoles afin de remplacer les engrais commerciaux. Cependant, des lacunes subsistent en matière d'informations sur l'impact de cette pratique sur le changement climatique. Une analyse de l'empreinte carbone a été réalisée pour quantifier l'impact sur le changement climatique associé au traitement et à l'épandage de trois types différents de biosolides : les biosolides digérés, compostés et alcalins. Les biosolides ont été épandus sur des terres agricoles de la ferme de recherche du campus Macdonald de McGill, près de Montréal, au Canada. OpenLCA 1.11 couplé à la base de données d'inventaire du cycle de vie Ecoinvent 3.6 ont été utilisés pour effectuer une évaluation de l'empreinte carbone de scénarios incluant chacune des trois méthodes de traitement afin de déterminer leur impact sur le réchauffement climatique. L'analyse comparative a révélé des résultats différents selon le scénario d'élimination par défaut (c'est-à-dire selon les émissions évitées prises en compte dans l'analyse). Dans le premier cas, les émissions évitées dues à l'élimination des boues n'ont pas été prises en compte. Dans ce cas, le scénario ayant le moins d'impact sur le changement climatique était l'application d'engrais à base d'urée, suivi de près par la digestion des biosolides. Dans le deuxième cas, si l'on considère les émissions provenant de l'incinération des boues, le scénario de traitement ayant le moins d'impact sur le changement climatique est l'épandage de biosolides digérés. Enfin, si l'on considère les émissions provenant de la mise en décharge des boues, le scénario ayant le moins d'impact sur le changement climatique est l'épandage de biosolides compostés. Les résultats mettent en évidence l'importance de détourner les matières organiques de la mise en décharge ou de l'incinération, ainsi que les émissions de gaz à effet de serre potentiellement associées ou évitées grâce au traitement et à l'épandage des boues d'épuration. Dans l'ensemble, une évaluation globale incluant les processus en amont et les émissions évitées est essentielle pour une comparaison équitable des différents scénarios de traitement et d'application.

Acknowledgements

Embarking on this research has been a transformative experience, enriched by the consistent support and contributions from various individuals and organizations.

At the forefront, I would like to express my deepest gratitude to my supervisor, Prof. Grant Clark, for his support and guidance throughout the course of this research. I am also thankful to Dr. Michael Boh, the project manager of the Agricultural Greenhouse Gas Project (AGGP2-033) for his insights and commitment to the project. I am grateful to have met Jing Ren, my lab mate, and appreciate her unwavering support throughout this journey.

My appreciation extends to the dedicated team at CIRAIG (Centre International de Référence sur l'Analyse du Cycle de Vie et la Transition Durable). Thank you, Prof. Anne-Marie Boulay, for introducing me to the world of LCA and being an advisor on my committee. I'd like to extend my gratitude to Dominique Maxime for his invaluable assistance in initiating the collaboration on this project. I am particularly indebted to Pablo Tirado. His assistance with the LCA modelling, combined with his patience and expertise, has been instrumental in the realization of this project.

I would also like to acknowledge the generous financial support provided by the Ministry of Agriculture and Agri-Food Canada. Their backing through the Agricultural Greenhouse Gas Program has been essential in facilitating this research.

On a personal note, my heartfelt thanks go to my parents, Fouad and Patricia. Through their actions and guidance, they've instilled in me the principles of resilience and adaptability. Thank you to my siblings, Maëlle and Joey, for having been part of this journey with me. I'd also like to extend my gratitude to longstanding friends and those I've met along the way. Their unwavering support has been invaluable. To all who have been a part of this journey, directly or indirectly, thank you. Your contributions, big and small, have made this thesis possible.

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Nomenclature

Acronyms	**Full Meaning**
AAFC	Agriculture and Agri-Food Canada
AD	Anaerobic Digestion
AGGP	Agricultural Greenhouse Gas Program
ATB	Alkaline Treated Biosolids
BEAM	Biosolids Emissions Assessment Model
CCME	Canadian Council of Ministers of the Environment
CEPA	Canadian Environmental Protection Act
CO_2e	Carbon Dioxide Equivalent
COM	Composting
CRAAQ	Centre de référence en agriculture et agroalimentaire du Québec
CVMO	Centre de Valorisation des Matières Organiques
ECCC	Environment and Climate Change Canada
GHG	Greenhouse Gas
GWP	Global Warming Potential
INC	Incorporated (method of biosolids application)
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
SOC	Soil Organic Carbon
SS	Surface-Spread
UF	Urea Fertilizer
UN	United Nations
WWTP	Wastewater Treatment Plant

Chapter 1 Introduction

Since the early 1970s, humankind has been driving the planet into a global ecological overshoot (World Wildlife Fund, 2022).With rapid urban expansion, the global society of today is consuming more resources and energy than the planet can regenerate in a year and producing more waste than it can assimilate. Today, humanity uses the equivalent of 1.75 Earths to provide resources and absorb waste (World Wildlife Fund, 2022). If global consumption trends continue on a business-as-usual, humanity will be using the equivalent of two planets to meet the demand of an expected population of 8.5 billion people by 2030 (UN, 2023). A global overshoot in resource consumption requires an imminent end to the era of infinite expansion and the start of an era threatened by severe limits and scarcity.

As our world faces growing threats from resource scarcity and climate change, it is critical that we transition away from the traditional linear economy model and move towards a circular economy that maximizes the use of natural resources in a "closed-loop system". The circular economy is based on the idea that products and resources should be kept in the economy for as long as possible, with waste being viewed as a secondary resource that can be recycled and reused (Neczaj & Grosser, 2018). Unlike the linear economy model, where waste disposal is the final stage in a product's life, the circular economy offers numerous opportunities for repurposing waste products in various sectors (Neczaj & Grosser, 2018). By embracing this circular approach, we can create a more sustainable future by reducing waste, conserving resources, and minimizing our impact on the environment.

One area where the circular economy model can be especially impactful is in the urban wastewater treatment sector. Among the pressing challenges in wastewater management today, sewage sludge handling stands out as a significant concern, given its status as the largest by-product generated from wastewater treatment processes. Wastewater treatment facilities across the world produce enormous amounts of sewage sludge that must be treated. Over the past few years, its management in a manner that is both economically feasible and environmentally acceptable has gained importance (Yoshida et al., 2013). Municipal sewage sludge generation is expected to further increase due to many factors. First, Canadian regulations and guidelines have been increasingly prioritizing wastewater treatment, thereby amplifying sewage sludge production. Remarkably, over 69% of Canadians are now served by municipal sewers equipped with secondary or higher levels of treatment, thanks to the modernization of the wastewater treatment sector

(Environment and Climate Change Canada, 2023). As the sector improves, a larger influent flow enters wastewater sewage volumes, leading to the removal of greater total solids during the treatment process. Additionally, stringent effluent discharge standards imposed by federal, provincial, and municipal governments necessitate compliance by wastewater treatment plants (WWTPs) to safeguard aquatic environments. In 2009, the Canadian Council of Ministers of the Environment (CCME) endorsed a nationwide strategy to manage municipal wastewater effluent from more than 3,500 facilities in a harmonized and environmentally protective framework (Sylvis, 2009). Consequently, the implementation of this strategy is expected to result in the construction of new and upgraded WWTPs, leading to an anticipated increase in the production of municipal sludge. Consequently, WWTP operators are now faced with the challenge of managing escalating quantities of sewage sludge.

On the other hand, sewage sludge, previously seen as a waste to disposed of in landfills, is now increasingly being recognized as a valuable resource. With growing concerns about resource scarcity, there has been a shift in perception towards viewing sewage sludge as a potential asset rather than mere waste. Sewage sludge contains valuable crop nutrients such as nitrogen (N), phosphorus (P) and potassium (K) along with organic matter that can improve soil conditions if applied on agricultural lands. However, before any disposal or management steps, further treatment is necessary to mitigate potential risks to human and environmental health. This treatment process aims to diminish odor, reduce, or eliminate pathogens, and prevent attraction of potential vectors to the sludge. Upon successful treatment and meeting federal and provincial government requirements for safe use, the sewage sludge is transformed into a stabilized product hereafter referred to as "biosolids". Biosolids can be used beneficially in other sectors (e.g., land-applied), stored or disposed without further usage in landfills or are incinerated.

Handling large quantities of biosolids is an ongoing and inevitable challenge in wastewater management. The disposal of sewage sludge in landfills was a common practice for wastewater plant operators in the past but there has been a significant shift away from this approach due to associated greenhouse gas (GHG) emissions including menthane and carbon dioxide as well as groundwater leaching. For instance, of the 44 Gt carbon dioxide equivalent (CO_2e) of global anthropogenic emissions in 2005, landfill emissions represented ~2% (794 Mt. CO_2e) and are increasing by 1–2% annually (UN-HABITAT, 2008)

In addition, with the aim of promoting nutrient recovery and offsetting GHG emissions from landfills, some Canadian jurisdictions such as Ontario and Nova Scotia have implemented the ban of organic matter from landfills (2020 and 1997 respectively) thus diverting biosolids from landfills and promoting resource recovery and contributing to circular economy.

With arising regulations that limit the discharge of organic matter in landfills around Canada, wastewater plant operators are diverting from the practice of landfilling sewage sludge as a management option. This implies there is pressure for reducing the share of sewage sludge going to landfill. With the above-mentioned arguments, it can be expected that, there will be a growing interest to treat sewage sludge to produce biosolids, an organic based product stabilized to meet jurisdictional standards. Municipal biosolids can be beneficially utilized as soil amendment and fertilizers on agricultural lands to produce crops. The recycling of biosolids on agricultural lands not only diverts sewage sludge from landfills, but also achieves the recovery of nutrients by recycling them to the soil.

According to Environment and Climate Change Canada (ECCC), land application of treated municipal sludge or biosolids is a beneficial option in contrast to other disposal options such as landfilling. Land application of biosolids has various benefits in terms of soil productivity and resource recovery (Environment and Climate Change Canada, 2023). Considering the documented benefits of the practice of biosolids land application, the treatment of wastewater sewage sludge has been increasing steadily, remarkably in the United States, Canada, the United Kingdom, and Australia where land application of biosolids is the most common beneficial reuse pathway (Alvarez-Gaitan et al., 2016). In Canada, biosolids production has increased from an estimated 554,000 dry tons in 2001 to 780,000 dry tons in 2015. In 2015, 53% of the total biosolids produced by Canadian WWTPs have been land-applied to crop lands and 12% were landfilled (Cheminfo Services Inc., 2017). The use of biosolids on agricultural lands is thus the most common disposal and management option for biosolids in Canada. Yet, Canadian jurisdictions still have a need to further optimize nutrient recycling from biosolids land application. With steadily increasing quantities of biosolids being applied to agricultural lands, some municipalities lack a sound management strategy to manage the large volumes while taking into consideration site-specific climatic conditions and soil characteristics. Sustainable nutrient management requires that we find the best combination of biosolids treatment and application techniques to optimize nutrient recovery while minimizing environmental impacts particularly, GHG emissions.

In order to support the sustainable growth of the Canadian agricultural sector, the federal government ministry of Agriculture and Agri-Food Canada (AAFC) developed a funding program entitled Agricultural Greenhouse Gases Program (AGGP). This program provided financial support to projects that created innovative technologies, practices and processes that can be adopted by farmers to mitigate GHG emissions (AAFC, 2018). The AGGP funded projects will also help farmers increase their understanding of GHG emissions. The research project entitled "***Management strategies for nutrient use efficiency and GHG emissions reduction from biosolids-amended soils in Canada***" aligned with AGGP's goals and purposes. The project was submitted for application to AGGP targeting Priority Area B: Cropping systems (Fertilizer Use Efficiency). The project was granted the AGGP2-033 funding in November 2016 and is expected to be completed in March 2020. The project's goal was to investigate management practices for use of municipal biosolids as a crop fertilizer and to quantify the effect of these practices on GHG emissions and crop nutrient use efficiency. The overall objective of this project was to assess the effect of biosolids pre-treatment and application methods on N use efficiency, GHG emissions, and C sequestration, under three distinct climatic conditions in Canada: Humid continental (Montreal), Atlantic maritime (Nova Scotia) and semi-arid Prairie (Edmonton). Thus, the experimental project has three pseudo-replicates respectively in McGill University (Montreal), Dalhousie University (Nova Scotia) and University of Alberta (Edmonton). This research was based only on the Montreal experiment that took place at the Emile A. Lods Agronomy Research Centre, Ste-Anne-de-Bellevue, Quebec, Canada (Latitude 45°28′N, Longitude 73°45′W) and lasted for three years (2017-2019).

The objective of the biosolids project involved extensive GHG emissions measurements from biosolids-amended soils to quantify the impacts of biosolids application on climate change. However, the quantification of GHG emissions from land-application isn't enough information to support the decision of policy makers regarding the best biosolids management strategy. The scope of study needs to be further expanded to include upstream processes such as different biosolids treatment processes, their storage and transportation. Different biosolids treatment processes require varying energy and chemical inputs (Brown et al., 2010). Therefore, the overall environmental impact cannot be fully assessed unless potential emissions associated with the full range of biosolids management options are included. Many studies have focused on assessing the environmental impacts of different treatment and end uses options, but few have captured the holistic assessment of biosolids management from production to land application in Canadian climate and soil conditions.

The results of this thesis will be of interest to three groups of stakeholders. The agronomic aspect of that project will concern the agricultural producers or farmers as they are interested in the potential of having fertilizer offsets. The aspect of waste disposal or waste valorization will be of interest to wastewater plant operators. Finally, the reporting of GHG emissions associated with biosolids land-application land will aid government and policy makers in the accounting of GHG emissions from the agricultural sector. Particularly, these new site-specific data will aid Environment and Climate Change Canada (ECCC) to optimize models such as the Biosolids Emissions Assessment Model (BEAM) to better quantify GHG emissions associated with biosolids land application. The improvement of GHG calculating tools results in a more comprehensive GHG monitoring making it possible for identifying effective climate change reduction strategies on the corporate, jurisdictional, or governmental level.

The goal of this research is to quantify the GHG emissions of different biosolids management options in Quebec Canada and compare them to determine a best preferred scenario of practice based on global warming potential.

The research question is the following:

Which biosolids treatment and application technique would best limit GHG emissions from the agricultural sector in Canadian climate and soil conditions?

The main objectives of this study are:

1- To design a comparative carbon footprint assessment of different scenarios of biosolids production technologies and application methods
2- To quantify GHG emissions associated with each scenario.
3- To determine the Global Warming Potential (GWP)
4- To provide recommendations for farmers, WWTP operators and climate policy makers on how to limit agricultural emissions based on our results.

Chapter 2 Literature Review

2.1. General Overview

The fundamental purpose of all wastewater treatment operations is to remove undesirable constituents present in wastewater sewage and stabilize these materials for further handling and disposal (Shammas & Wang, 2008). In Canada, the government takes responsibility in protecting the environment and human health by acting on pollution prevention and monitoring of wastewater treatment effluents quality. Examples include the Canadian Environmental Protection Act (CEPA) established in 1999 (Government of Canada, 1999) and the Fisheries Act (Government of Canada, 1985). The first act aims for the protection of the Canadian environment and human health by focusing on the prevention and management of risks posed by the usage of toxic and other harmful substances The latter includes wastewater systems effluent regulations which represent mandatory minimum effluent quality standards for wastewater plants that discharge treated effluents into water bodies inhabited by fish.

In recent decades, urbanization and modernization of the wastewater treatment sector have led to increasing volumes of wastewater being treated. In 2017, WWTP facilities across Canada have received just over 5900 m^3 of sewage every day (Statistics Canada, 2019). Typically, incoming sewage is treated through physical, chemical, and biological processes. The primary treatment stage removes bulky solids while the secondary treatment is a biological process. During that stage, microbial biomass is flocculated by the addition of chemicals such as aluminum sulphate or calcium hydroxide. The flocculated solids settle to the bottom of the tank and is removed mechanically with a skimmer (Metcalf et al., 2003). The primary and secondary treatments of sewage produce primary and secondary sludge part of which is recycled back into the secondary treatment and the rest remains to undergo further stabilization. It is thus clear that an inevitable by-product of the wastewater treatment industry is sewage sludge. Sewage sludge is a semi-solid and slurry waste that is formed during the preliminary, primary, secondary, and tertiary treatment of wastewater. Sewage sludge is a combination of suspended, and dissolved solids removed from residential, institutional, commercial and industrial sectors and contains organic, inorganic, harmful chemicals as well as pathogenic microorganisms. Sewage sludge in its untreated form can be harmful to human and environmental health and therefore must undergo further treatment before any disposal. Once the sludge is properly treated and tested to meet government requirements for beneficial use, it can be termed “biosolids”.

There are many definitions of the term “biosolids” and these can vary slightly in the literature depending on the context and the source of information. For example, The United States Environmental Protection Agency defines biosolids as "treated sewage sludge that meets specific criteria for land application and surface disposal" (Boczek et al., 2023). It refers to the residual solids that remain after wastewater treatment processes. On the other hand, the CCME defines biosolids as "the nutrient-rich organic material resulting from the treatment of sewage sludge from municipal WWTPs” (CCME, 2012). This definition specifically refers to sewage sludge from municipal WWTPs. These definitions highlight the common aspects of biosolids, such as their origin from sewage sludge or wastewater treatment, their nutrient-rich nature, and the treatment processes they undergo to ensure their safety and suitability for beneficial use. As such it can be highlighted here that definitions may vary slightly across different countries, regions, and regulatory bodies. In the context of this thesis, the term “biosolids” will refer to stabilized municipal sewage sludge derived from municipal wastewater treatment processes which can be managed safely to be used beneficially for their nutrient, soil conditioning, energy generation and other values (Shammas & Wang, 2008).

Biosolids are organic materials derived from the treatment of wastewater and represent a good source of organic matter ranging from 50–70% (Wijesekara et al., 2016). They can be used as a fertilizer or soil amendment due to their nutrient content. The specific nutrient composition of biosolids can vary depending on the treatment process and the source of the wastewater as well a geographical locations and seasons (Arulrajah et al., 2011). However, generally, biosolids contain the following nutrients:

- Nitrogen (N): Biosolids are a significant source of nitrogen. The nitrogen content can range from 1–6%, and it exists in various forms, including organic nitrogen, ammonium, and nitrate (Brown & Henry, 2001).
- Phosphorus (P): Phosphorus is an essential nutrient for plant growth and development. The phosphorus content in biosolids is typically around 0.7–7.5% (Kim & Owens, 2011).
- Potassium (K): Biosolids can contain potassium, which is necessary for plant health. The potassium content in biosolids is usually around 0.1–0.6% (Dan M. Sullivan et al., 2015). In

most situations, the provision of K from biosolids is insignificant and fertilizer must be supplied to meet the plant requirements (Kim & Owens, 2011).

- Micronutrients: Biosolids may contain trace amounts of micronutrients such as iron, manganese, copper, zinc, and molybdenum. These micronutrients are essential for various biochemical processes in plants (Dan M. Sullivan et al., 2015).

Biosolids can potentially contain harmful substances due to the presence of contaminants that enter the wastewater treatment system. These contaminants can come from various sources, including industrial discharges, household chemicals, pharmaceuticals, and other substances that are disposed of through wastewater. The types and concentrations of harmful substances in biosolids can vary depending on factors such as the source of wastewater, the treatment processes employed, and the level of control and monitoring in place. Some of the potentially harmful substances that can be present in biosolids include pathogens, heavy metals, organic chemicals, Endocrine-disrupting compounds (EDCs) and other emerging pollutants.

Biosolids can harbor pathogens such as bacteria, viruses, and parasites that may be present in wastewater. These microorganisms can pose risks to human health and the environment if not properly treated or managed (CCME, 2012; Sablayrolles et al., 2010).

Certain heavy metals can enter wastewater through industrial processes, runoff from roads, or household discharges. Examples of heavy metals that can be found in biosolids include lead, mercury, cadmium, arsenic, and chromium. These metals can have toxic effects on human health and the environment if present in high concentrations (Cheminfo Services Inc., 2017).

Biosolids may contain organic chemicals such as pesticides, herbicides, industrial chemicals, pharmaceuticals, and personal care products. These substances can be introduced into wastewater through various sources and may persist in the biosolids. Some organic chemicals can have adverse effects on human health and the environment (CCME, 2012).

In addition, EDCs are chemicals that can interfere with the hormonal system of humans and wildlife. Some EDCs can enter wastewater and potentially accumulate in biosolids. Examples of EDCs include certain pesticides, plasticizers, and pharmaceuticals.

Finally, the COVID-19 outbreak led to an increased use of personal care products and partially metabolized antibiotics, which resist wastewater treatment processes and end up being adsorbed onto the sludge fraction in significant amounts compared to normal conditions (Rizvi & Ahammad, 2022).

To mitigate potential risks associated with harmful substances in biosolids, strict regulations and guidelines are in place in many jurisdictions: e.g., Canadian Environmental Protection Act (CEPA) (Government of Canada, 1999), Fisheries Act (Government of Canada, 1985), as well as the guidance document for the beneficial use of municipal biosolids, municipal sludge and treated septage (CCME, 2012). These regulations and guidelines specify acceptable limits for various contaminants and outline procedures for monitoring, treatment, and safe use or disposal of biosolids.

2.2. Biosolids Treatment Techniques

Stabilization of sewage sludge can be achieved chemically, biologically, or thermally. The treatment of sewage sludge achieves the reduction of odor potential by reducing volatile organic compounds, a decrease or elimination of pathogen concentration and reduction in attraction of potential vectors to the sludge (CCME, 2012). In Canada, there are several treatment technologies that are currently popular for the processing and management of biosolids. The choice of technology depends on factors such as the size of the WWTP, the desired level of treatment, regulatory requirements, and the intended use or disposal method for the biosolids. Some of the common treatment technologies for biosolids in Canada include biological stabilization such as anaerobic digestion and composting as well as chemical treatments such alkaline stabilization. The treatment technologies considered in this study will be described in more detail in the following sections.

2.2.1. Anaerobic Digestion

Anaerobic digestion involves the breakdown of organic substances by microorganisms in an oxygen-deprived environment (Metcalf et al., 2003). The most prevalent form of this process is mesophilic anaerobic digestion, where sludge is maintained at temperatures between 30-38°C for a duration of 15 to 30 days within an oxygen-free vessel known as a digester (Metcalf et al., 2003). This reactor witnesses a series of simultaneous biochemical reactions, namely hydrolysis, acidogenesis, and methanogenesis, facilitated in a continuously stirred tank reactor.

During hydrolysis, organic compounds such as cellulose, lignin, lipids, proteins, and simple sugars are transformed into carbon dioxide (CO_2), alcohols, soluble fatty acids, and ammoniacal compounds (Leite et al., 2023; Sylvis, 2009). Subsequently, in the acidogenesis phase, these by-products are further degraded into low molecular weight organic acids, primarily acetic and propionic acids, along with hydrogen and CO_2 (Forster-Carneiro et al., 2010; Metcalf et al., 2003). The concluding phase sees the conversion of hydrogen and CO_2 into methane by methanogenic bacteria, while acetic acid is transformed into methane and bicarbonate by acetogenic bacteria (This procedure not only reduces the pathogen content in the sludge by approximately 95% but also significantly diminishes the odor of the resultant digestate.

The outcome of anaerobic digestion is digested sludge, called digestate and a co-product which is biogas. Biogas can be used beneficially as a renewable source of energy which makes anaerobic digestion an appealing choice of treatment. Overall, 40–60% of the organic solids are converted to biogas of which 60–65% is methane and 30–35% is CO_2 (Brown et al., 2010). The biogas also contains traces of H_2, N_2, H_2S and H_2O. The biogas can then be used for many beneficial uses such as in combustion to run generators producing electricity, used as a fuel in furnaces or cooking stoves, or purified as a replacement for natural gas (Niu et al., 2013; Sylvis, 2009). The produced biogas is however most often used for process heating and electricity production.

As an example, the city of Saint-Hyacinthe (Quebec) has been leading in the field of biomethanisation realizing the anerobic digestion of the region's sewage sludge and organic matter to produce 13 million m^3/year of biogas (Ville de Saint-Hyacinthe, 2023). The captured biogas from the anaerobic digesters is used to power municipal vehicles and provide heating and cooling for the city's buildings. The surplus biogas is sold to Énergir, the regional public utility. By utilizing the anaerobic digestion process instead of disposing of the sewage sludge in landfills, a municipality can in turn reduce their emissions of landfill gases and the WWTP can offset some of its GHG emissions by reducing its consumption of natural gases for running its operations.

2.2.2. Alkaline Stabilization

The process of alkaline stabilization consists of adding alkaline chemicals to raw sewage sludge to raise the pH to 12 or higher. High pH and appropriate mixing and contact time stops or considerably decelerates the reactions of microorganisms that can otherwise lead to production of odor and attraction of different vectors (e.g., rodents and insects) (Metcalf et al., 2003). Alkaline additives commonly used include lime in the form of (Ca(OH)2) and quicklime (CaO) (Metcalf et al., 2003). Quicklime is frequently favored due to its substantial heat of hydrolysis (6.25 kJ/g) (Zumdahl, 2009) which greatly helps in pathogen elimination. Other product variations can also be used.

One example of an alkaline stabilization method is the N-Viro process which is a patented process for the treatment and recycling of bio-organic wastes, utilizing certain alkaline by-products such as cement kiln dust, lime kiln dust, fly ash and steel-making fines (N-Viro Systems Canada inc., 2007). If the alkaline admixture does not contain enough free lime or other strong alkali) to give the necessary temperature and pH rise, quicklime (CaO) is added. Cement kiln dust is commonly utilized for alkaline stabilization due to its ready availability and relatively affordable cost. Cement kiln dust is generated during the production of cement in kilns. Throughout the cement manufacturing process, various dust collection systems are employed to capture and collect particulate matter generated. This collected dust is typically composed of partially calcined and unreacted raw materials made of calcium oxide (CaO), which contributes to its alkaline properties (Sylvis, 2009).

In the N-Viro Soil patented process an alkaline admixture is added to dewatered sludge in a mixing bin. Typically, 30–40% of the mixture is added on a biosolids wet weight basis (N-Viro Systems Canada inc., 2007). The discharge from the mixer travels by conveyor directly into the mechanical rotary-drum where it is dried to 60–65% solids content. A combination of heat from the dryer and further chemical reaction between the alkaline materials and the biosolids maintains the temperature within a controlled range of 52–62 °C, and the pH slightly greater than 12. As in the mixing stage, this combination of heat and high pH in this step is important in the destruction of harmful pathogens. The material discharged from the dryer proceeds to a "heat- pulse cell" where the material is cured for twelve hours. The heat-pulse cell contributes to stabilization of the product and pathogen kill. The process reduces odours to acceptable levels, neutralizes or reduces the mobility of some heavy metals, and generates a product that has a granular appearance similar to soil. A notable characteristic to alkaline stabilization is that it will result in slightly increased volumes of wastewater residuals requiring management due to the addition of the alkaline admixture (CCME, 2012). According to the achieved level of stabilization, the alkaline-stabilized sludge, now referred to as alkaline biosolids, can possibly be beneficially reused as a fertilizer and soil conditioner, cover for solid waste in landfill etc.

2.2.3. Composting

Composting sewage sludge is another widely practiced method of stabilization in Canada. Composting involves the biological decomposition and stabilization of organic substrates. This process generates thermophilic temperatures (around 45°C) as a result of biologically produced heat, resulting in a final product that is stable, free of pathogens and can be beneficially applied to land (Haug, 1993). Composting can be anaerobic or aerobic. Anaerobic composting is the biological decomposition of organic substrates in the absence of oxygen whereas aerobic composting takes place in the presence of oxygen. Municipal sewage sludges are often composted by adding amendments such as sawdust, straw, yard waste as well as food waste (Haug, 1993).

Composting processes can be broadly categorized into two types: reactor and nonreactor systems. In reactor systems, the composting material is housed within a specific reactor, often referred to as "in-vessel" processes. Conversely, nonreactor systems, commonly known as "open" systems, do not utilize a reactor for the composting material (Haug, 1993). The windrow system stands out as a prominent example of a nonreactor, agitated solids bed system. In this approach, mixed feedstocks are arranged in rows and are periodically turned, typically using mechanical tools. When composting sewage sludge by windrow processing, the biosolids are converted to a relatively stable organic residue and reduced in volume by 20–50%. The residue loses its original identity with respect to appearance, odor, and structure. The end product has earthy characteristics, while pathogens, weed seeds, and insect larvae are destroyed (Haug, 1993).

Composting produces a stable product high in organic matter which can be used as a soil amendment. To a lesser extent, compost can provide a source of nutrients; however, compost typically exhibits lower nutrient concentrations compared to other types of biosolids, a consequence of microbial consumption during the composting process (Haug, 1993). Lundin et al. (2000) and Poulsen and Hansen (2002) assumed losses of 50 and 33% of nitrogen through denitrification and ammonia volatilization respectively.

2.3. Biosolids End-Use and Disposal Options

Biosolids production has been steadily increasing in Canada over the years. In 2001, it was estimated that 554 000 t of dry biosolids were produced across the country (Cheminfo Services Inc., 2017). This volume increased to reach 780 000 dry tons in 2015. The increase in biosolids production and treatment can be attributed to several factors (Yoshida et al., 2013). First, wastewater plant operators are being faced with increasingly stringent nutrient discharge standards which causes the removal of more organic matter and other unwanted particles from the wastewater. Second, there are potentially over 100 landfills in Canada that accept municipal wastewater treatment sludge/biosolids (Cheminfo Services Inc., 2017). However, several factors such as regulatory influences, voluntary improvements in biosolids quality, and the resulting increase in biosolids use are leading to the declining amount of sludge/biosolids that are disposed of in municipal landfills in Canada. The banning of organic waste from landfills is becoming increasingly popular in Canada. For example, such a policy was implemented in Nova Scotia starting in 1997 to promote the recovery of municipal residuals (Government of Nova Scotia, 1996). Ontario plans to ban food and organic waste from ending up in disposal sites phased-in beginning 2022 under the food and organic waste framework (Govenment of Ontario, 2017). Finally, the increase in compliance costs is making it more expensive for WWTP operators to dispose of their sludge in landfills. An example of such cost is the green taxing in Quebec that represents a tax of $19.50 for each ton of sludge that is landfilled (Cheminfo Services Inc., 2017).

Because of these regulations, WWTP operators are being driven to consider other ways to dispose of biosolids in a more sustainable fashion. The handling of biosolids is one of the most significant challenges in wastewater management (Metcalf & Eddy, 1991). This is because there are benefits, risks and specific considerations for every management option. Municipal biosolids management options depend on the characteristics and quality of the municipal biosolids, the treatment process used to produce the municipal biosolids and the legislative framework of the province or territory where the municipal biosolids are used. Nevertheless, the are five main management option categories for biosolids in Canada (CCME, 2012; Cheminfo Services Inc., 2017):

- Landfilling
- Incineration
- Land application for crop production
- Forestry or reclamation of waste land
- Storage

According to Canadian Council of Ministers of the Environment (CCME), the management options of biosolids can be classified in two categories: Beneficial use options and disposal options: Beneficial use options capitalize on the nutrient and organic matter value and energy content of the municipal biosolids and are used for energy production through combustion, compost and soil products, agricultural land application as a fertilizer or soil conditioner, forestry application as a fertilizer or soil conditioner and land reclamation (CCME, 2012). Disposal options include those that do not have a consideration of the utility and resource value of biosolids; for example, burying municipal biosolids in a landfill or combustion without energy recovery. Landfilling municipal biosolids is not a beneficial option compared to other methods. The decomposition of organic matter in landfills contributes to methane emissions and groundwater contamination due to leaching (Cheminfo Services Inc., 2017). Several studies such as Suh & Rousseaux (2002) and Houillon & Jolliet (2005) who have studied the landfilling of sewage sludge, have reported considerable methane emissions by this practice, and have suggested its avoidance by re-using the sewage sludge as fertilizer (Houillon & Jolliet, 2005; Suh & Rousseaux, 2002). Canada generates approximately 780,000 dry tons of sewage annually. The management of this sludge/biosolids was as follows: 53.4% to land application (416,492 tonnes); 28.8% to incineration (224,801 tonnes); 12.1% to landfills (94,784 tonnes); 3.8% to storage (29,563 tonnes); and1.9% to land reclamation (14,737 tonnes) (Cheminfo Services Inc., 2017) (CCME, 2012).

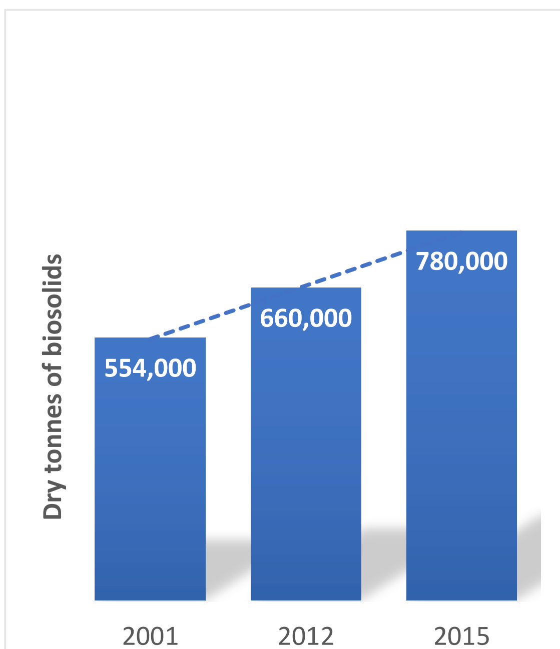

Figure 2.1. Annual production of biosolids in Canada (CCME, 2012; Cheminfo Services Inc., 2017)

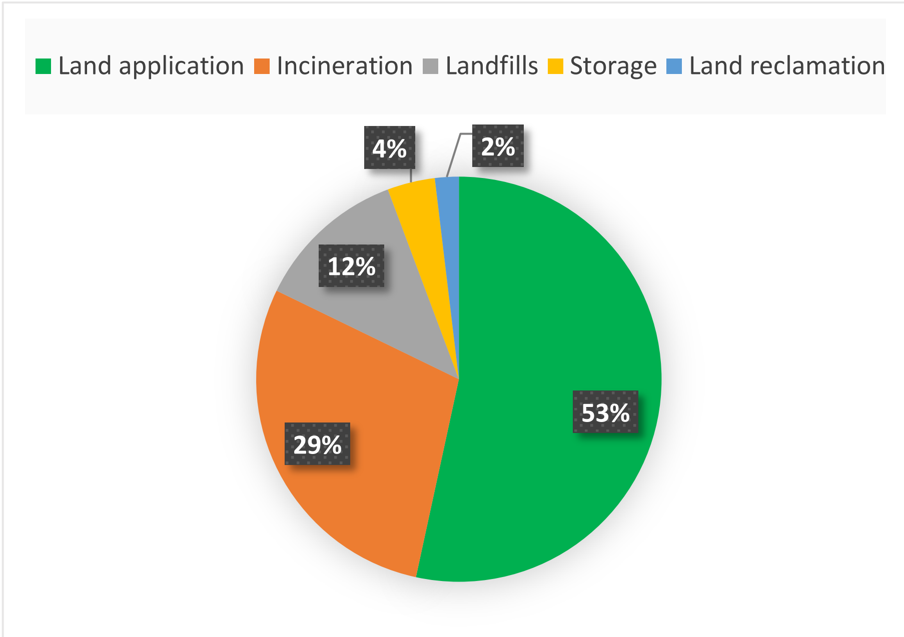

Figure 2.2. Biosolids management in 2015 in Canada (Cheminfo Services Inc., 2017)

2.4. Land Application of Biosolids

In contrast to landfilling, the management of biosolids by land application has been gaining more popularity due to the acknowledged benefits of this practice. The use of biosolids in agriculture is, however, not a novel concept. For centuries, human waste has been applied to the land as a mean for fertilizing the soil in many parts of the world such as China and Europe. Today, land application of biosolids in still the most common beneficial reuse pathway in countries such as the United States, Canada, the United Kingdom and Australia (Spinosa, 2011).

Land application of biosolids has various benefits in terms of soil productivity and resource recovery. Municipal biosolids contain up to 50% organic carbon as well as essential macronutrients (e.g., nitrogen, phosphorus and potassium, calcium, magnesium, and sulfur) and micro-nutrients (e.g., boron, cobalt, copper, iron, manganese, molybdenum, nickel, and zinc), and can be used as a fertilizer or soil conditioner when applied on agricultural soils (Carbonell et al., 2011; Evanylo et al., 2006). The use of biosolids in agriculture has been shown to improve soil health (Brown et al., 2020; Nicholson et al., 2018; Wang et al., 2008). Nicholson et al. (2018) and Wang et al. (2008) have reported an improvement in soil structure, soil porosity, water holding capacity of soils resulting from the application of biosolids. In return, the addition of organic matter helps maintain soil health and reduces the potential for soil erosion (Casado-Vela et al., 2006; Nicholson et al., 2018; Ojeda et al., 2003). When biosolids are properly applied, farmers can expect an increase in crop yield due to improved soil fertility (Boudjabi & Chenchouni, 2021; Sullivan, 2015; Wang et al., 2006).

Besides improving soil quality, biosolids application can supplement or replace commercial fertilizers such as urea. Nutrients such as nitrogen in biosolids are less water-soluble in their organic form (Wang et al., 2009). As soil bacteria slowly process through decomposition, N and other nutrients are slowly released over several growing seasons thus creating a residual effect over the years. A study conducted by Binder et al. (2002) reported that approximately 40, 20, 10, and 5% of the total biosolids-N were recovered by the crops in the 1st, 2nd, 3rd, and 4th year, respectively, after a single biosolids application. The relative yield increase was 33%, 21%, 14%, and 9% in the 1st, 2nd, 3rd, and 4th year, respectively, after application (Binder et al., 2002). The slow release of nutrients is more beneficial to crops as these nutrients remain in the cycle over time and are less likely to be leached. In contrast, most nutrients in commercial fertilizers are water soluble, readily subjected to leaching losses if they are not rapidly taken up by the crops (Sampson, 2016).

The application of municipal biosolids on crop lands achieves a complete recovery of N, P and K. This closed-loop system could reduce the need for additional soil nutrients or amendments for crops since the supply can be covered by biosolids application to the land. In addition, due to the durability of nutrients in biosolids in the soil, farmers are expected to use lower amounts of commercial fertilizers to supplement nutrients supplied through biosolids. Also, the improvement in soil quality and increase in crop yield over time will result in an increase in crop revenue for the farmers. As a result, land application of biosolids can potentially substitute the use of commercial fertilizers (e.g., Urea) and thereby avoids energy and resource-intensive manufacture of commercial fertilizers with its associated GHG emissions (Willén, 2016).

Furthermore, phosphorus recovery has been the main driver for biosolids land application and has been the focus of several paper studies on phosphate recovery to agricultural lands (Johansson et al., 2008; Lederer & Rechberger, 2010; Linderholm et al., 2012; Lundin et al., 2000). Nowadays, modern agriculture relies heavily on phosphorus-based fertilizers to sustain global food production. These fertilizers contain phosphorus derived from phosphate rocks, a finite and non-renewable resource that could be exhausted in the next 50 to 100 years at current extraction rates (Cordell et al., 2009). With a population expected to reach 9.8 billion by 2050 (United Nation, 2017), coupled with an impending phosphorus scarcity, strategies for sustainable phosphorus use and management should be implemented to reduce phosphorus demand. Land application of biosolids thus plays an important role in solving the global phosphorus depletion by recycling phosphorus from human excreta into crop lands for food production. Land application of biosolids could reduce the need for phosphorus-based fertilizer production thus decelerating phosphate rock extractions.

Finally, carbon sequestration is another important benefit of land-applying biosolids. Biosolids are rich in organic matter, which contains carbon. When biosolids are applied to the land, they introduce a significant amount of organic matter to the soil. This organic matter acts as a carbon source, contributing to the buildup of soil organic carbon (SOC) over time. Higher SOC levels result in increased carbon sequestration as more carbon is stored in the soil.

There are several methods for land applications of biosolids. Biosolids can be spread on the soil surface using specialized spreading equipment such as manure spreaders or broadcast spreaders (United Stated Environmental Protection Agency, 1995). This method involves uniformly distributing the biosolids across the field. On the other hand, biosolids can be mechanically incorporated into the soil using tillage equipment such as plows, discs, or harrows. This method involves mixing the biosolids into the top layer of the soil. Incorporation helps reduce odor, improves nutrient distribution, and enhances soil-organic matter interaction. In the case of injection, this method involves creating slots or openings in the soil and placing the biosolids directly into these openings. Injection helps reduce odor, minimize surface runoff, and enhance nutrient availability by placing the biosolids in closer proximity to the root zone.

2.5. Life Cycle Assessment: A Methodological Framework

In today's increasingly environmentally conscious world, when it comes to making decisions about a certain process or product, Life Cycle Assessment (LCA) has emerged as a tool for evaluating its environmental impact and guiding sustainable decision-making. As defined in the International Organization for Standardization (ISO) 14040–14044 (ISO, 2006a, 2006b), LCA is the evaluation of the inputs, outputs and potential environmental impacts of a product, system, or service throughout its life cycle. It is a holistic environmental evaluation method that studies products, processes, and services from “cradle to grave” meaning that a LCA takes into consideration the environmental aspects throughout a product’s life cycle starting from raw material extraction, manufacturing, use phase, and end-of-life processes (ISO, 2006a). LCA has various areas of application that includes product development, public policymaking, process optimization, decision-making as well as marketing. A common goal behind many LCAs is the identification of the areas of potential amelioration in a specific field or sector (ISO, 2006b). LCA is very efficient in identifying opportunities to improve the environmental performance of products at various points of their life cycle. The evaluation of the system throughout its entire life cycle allows the determination of the most emitting procedures by highlighting the hotspots within a complex process chain. In addition, LCA is a valuable tool used by decision makers in firms, government, or non-government organizations to support their decision making and environmental optimization procedures (ISO, 2006b). Notably so, it allows the comparison between alternatives scenarios, contributes to the development and utilization of cleaner technologies, and allows the maximization of the material and waste recycling (ISO, 2006b). LCA is performed in accordance with the principles and framework of the ISO 14040 series. The concept of life cycle methodology is depicted in figure 2.1 and the description of the steps follows (ISO, 2006a).

2.6. Goal and Scope

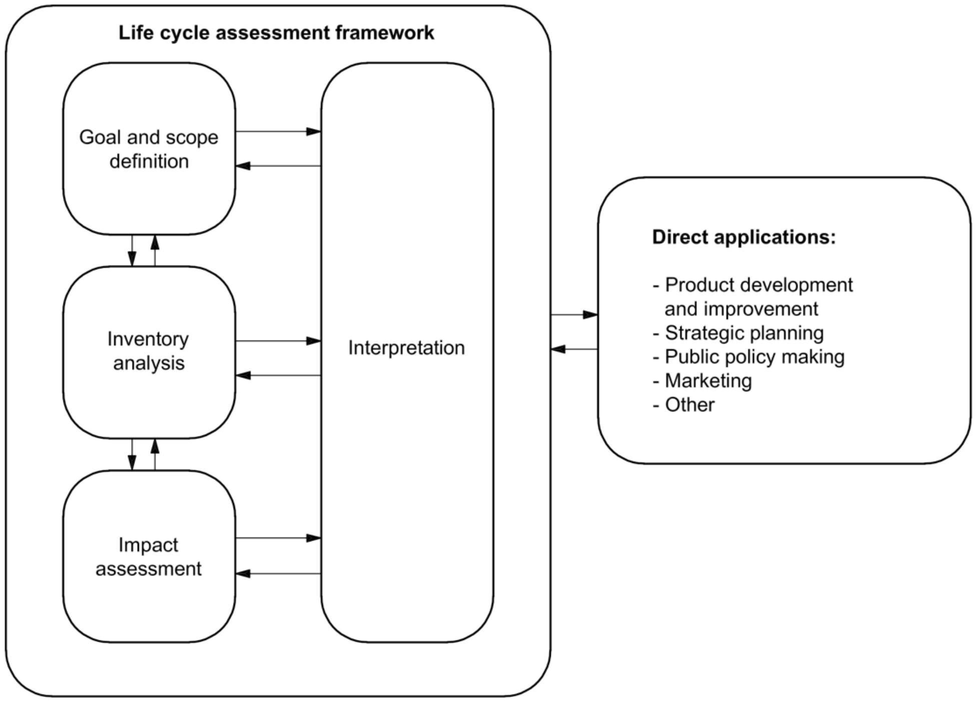

Figure 2.3. Methodological framework of an LCA as defined by ISO 14040 (ISO, 2006a)

The goal and scope of a Life Cycle Assessment (LCA) need to be precisely established and aligned with the intended purpose of the assessment. As LCA is an iterative process, this phase may be revisited and adjusted as necessary throughout the study.

a. Goal of study

During the goal definition phase in LCA, it is essential to clearly state the intended application of the LCA, outlining the specific purpose for conducting the assessment. Additionally, the reasons behind undertaking the study should be clearly articulated, highlighting the motivations and objectives driving the LCA. Moreover, the intended audience of the LCA should be identified, indicating the individuals or groups for whom the results are primarily intended. Lastly, it is essential to specify whether the LCA results will be used to make comparative assertions or if the goal is to disclose information to the public. This comprehensive goal definition ensures a well-defined framework for the LCA and enhances the clarity and relevance of the subsequent assessment.

b. Scope of study

The scope of the study established the system boundaries, data requirements and assumptions and limitations. It is essential to provide a detailed scope to ensure that the analysis aligns with the stated purpose and is comprehensive enough to address it. Clear documentation of data boundaries, methodology, data categories, and assumptions is necessary. This should include geographical context (local, national, regional, continental, or global) and the time frame (product life, time horizon of processes, and impacts).

i. Product system, Function and Functional Unit

The product or system process must be clearly presented as well as the performance characteristics of the system(s) being assessed. The functional unit must be consistent with the goal and scope of the study. The primary purpose of the functional unit is to provide a reference to which input, and output data can be normalized. Systems must be compared on the same iterative and comparative functional basis quantified by the same functional unit in the form of reference flows. By definition, a reference flow is a measure of the outputs from processes in each product system required to fulfil the function expressed by the functional unit.

ii. System boundary

The system boundary is designed as such that it fulfills the functional unit of the study. The system boundary determines the unit processes that are included with the LCA study. System boundaries must remain consistent with the goal of the study. The system can be described using a process flow diagram showing the unit processes and their interrelationship.

iii. Life Cycle Impact Assessment Methodology and Type of Impacts

In the Life Cycle Impact Assessment phase, it is important to determine the impact categories and establish category indicators and characterization models. The selection of these elements should be aligned with the study's objectives to ensure consistency.

iv. Types and Sources of Data

Collected data are dependent on the goals and scope of the study. Data can be collected from the production site associated with the unit process within the system boundary, sourced from peer reviewed articles in the literature or they can be measured, calculated, or estimated.

v. Data Quality Requirements

In LCAs data refers to the overall level of confidence in individual input and output data sets. When considering data quality requirements, various aspects need to be considered. These include aspects such as time-related coverage, geographical coverage, technology coverage, precision, completeness, representativeness, consistency, reproducibility, and the source of data.

2.7. Life Cycle Inventory

The life cycle inventory is the stage where all the data needed for modelling are to be collected. Its main purpose is to quantify the input and output flows that cross the product system boundaries (Jolliet, 2015). The life cycle inventory phase is an objective data-based process of quantifying energy and raw material requirements, air emissions, waterborne effluents, solid waste and other environmental releases incurred throughout the life cycle of a product, process or activity. This step involves compiling the inputs and outputs of each life cycle process associated with the product or service. It includes quantifying emissions and resources and providing descriptions of each unit process. The purpose of the inventory analysis is to list all substances emitted into or extracted from the environment during the product or service's life cycle.

Aspects of the life cycle inventory phase include:

- Data collection: detailed data in the form of input (material and energy) and outputs (product releases to air, water and land), data variability, uncertainty and sensitivity analysis. Sensitivity analysis is carried out to test the effects of the results and possible limitations on the conclusions.
- Allocation Procedures: when one or more useful output is produced in a subsystem, there arises the need for a consistent way to identify those inputs and outputs attributed to the system of interest in the study.

2.8. Life Cycle Impact Assessment

This is a technical, quantitative, and qualitative step to characterize and assess the effects of the environmental burdens identified in the life cycle inventory phase. This step includes the following elements (ISO, 2006a):

- Classification: Grouping of data in an inventory table into different impact categories
- Characterization: The quantification, aggregation, and analysis of impact data within the impact categories
- Weighting: Optional step that consists of multiplying the normalized results of each of the impact categories with a weighting factor that expresses the relative importance of the impact category. If the study focuses on one impact category, then the weighting step does not occur because there aren't several indicators to compare.

Impact categories are selected, and emissions are categorized accordingly. The substances identified in the inventory analysis are converted into environmental impacts based on cause-and-effect chains using impact assessment models. The computation can be conducted at the midpoint level (using incomplete cause and effect chains) or at the endpoint level (at the end of the chain).

Classification involves grouping of data in an inventory table into different impact categories, while characterization is the quantification, aggregation, and analysis of impact data within the impact categories. An impact category is a class representing environmental issues of concern to which life cycle inventory results may be assigned (ISO14044:2006). The selection process of the impact categories, category indicators and characterization models shall be both justified and consistent with the goal and scope of the LCA. A category indicator is a quantifiable representation of an impact category (ISO 14044:2006). Characterization models according the ISO14044 (2006) reflect the environmental mechanism by describing the relationship between the life cycle inventory results, category indicators and in some cases category endpoint(s). It is used to derive the characterization factors. The environmental mechanism is the total of environmental processes related to the characterization of the impacts. An example of the terms used the characterization process for climate change is presented in table 2.1 below.

Table 2.1. Characterization process in a life cycle impact assessment for climate change

Terms	Example
Impact category	Climate change
life cycle inventory results	Amount of GHG emissions per functional unit
Characterization Model	Baseline model of 100 years of the Intergovernmental Panel on Climate change (IPCC)
Category Indicator	Infrared radiative forcing (W/m^2)
Characterization Factor	GWP100 for each GHG (kg-CO_2e per functional unit)
Category indicator result	kg of CO_2 equivalents per functional unit

2.9. Interpretation

Interpretation is the last phase of an LCA. The results of the previous phases are interpreted to address the objectives set in the first step. Uncertainty and sensitivity analyses can also be performed during this phase. Conclusions, limitations, and recommendations are presented.

2.10. Co-Products in LCA

In addition to the principal product, a given system can generate one or more secondary products that have economic value, but do not correspond to the studied function. For example, the treatment of raw sewage sludge by anaerobic digestion generates anaerobically digested biosolids and biogas. But, since most LCAs focus on a single functional unit and, therefore, only on one product, we must somehow allocate impacts from a multiproduct system to a unique product. The LCA norms from the International Organization for Standardization (ISO) (ISO 14044:2006) define a hierarchy of allocation methods depending on the ISO denomination.

2.10.1. Allocation for Multifunctional Processes

Whenever possible, allocation should be avoided by

- dividing the unit processes to be allocated in two or more sub-processes and collecting the input and output data related to these sub-processes.

- Expanding the product system to include the additional functions related to the co-products.

In cases where allocation cannot be avoided, the inputs and outputs of the system should be shared between the different products or functions in a way that reflects the underlying physical relationship between them.

Where it is impossible to establish physical relationship or use it as a basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them (ISO14040). When there is no clear physical relationship to allocate resource use or emissions by-product, we consider economic causality, thereby capturing financial incentives. That is, a product is considered as primarily made for its mercantile value, so we can allocate emissions among coproducts according to their respective values (Jolliet, 2015).

2.10.2. Comparison between Carbon Footprint and LCA

To reiterate, an LCA provides a comprehensive evaluation of resource use, substance flows, and environmental impacts associated with a specific function. However, if the assessment solely focuses only on the effect of GHG emissions, it is referred to as a Carbon Footprint. A carbon footprint is simply the global warming component of an LCA and can be applied to products, activities, or companies. While an LCA examines how different scenarios can redistribute impacts across various impact categories, a carbon footprint concentrates solely on the greenhouse effect. By definition a carbon footprint measures the direct and indirect GHG emissions resulting from a product, human activity, or business (Jolliet, 2015). Basically carbon footprint calculation is also a life cycle approach; however, unlike LCA, it does not cover all emissions but only inputs that contribute to global warming (Taşeli, 2020)

2.11. LCAs on Biosolids Management

2.11.1. LCA in the Wastewater Treatment Industry

In the pursuit of sustainable sewage sludge management, the treatment of sewage sludge and disposal of biosolids have emerged as critical areas of focus. Biosolids, the nutrient-rich organic materials resulting from the treatment of sewage sludge, have the potential to be recycled and beneficially reused, contributing to the circular economy. However, the environmental impacts associated with different treatment and disposal options for biosolids are complex and multifaceted. Different biosolids treatment technologies require varying energy and chemical inputs (Brown et al., 2010) therefore the overall environmental impact cannot be fully assessed unless potential emissions and sequestration associated with the full range of biosolids management options are included from treatment to transportation to end-of-life.

Over the past years, a growing body of research has employed LCA methodologies to evaluate these impacts, providing valuable insights for policy makers, industry practitioners, and researchers (Corominas et al., 2013; Ding et al., 2021; Guinée et al., 2011; Yoshida et al., 2013). LCA has been extensively used in recent years in studies that have focused on assessing the environmental impacts of different treatment and end uses options for biosolids on a process level. Suh and Rousseaux (2002), Houillon and Jolliet (2015) and Sablayrolles et al. (2010) conducted an LCA in the aim of comparing conventional technologies for the treatment of sewage sludge. Murray et al. (2008) and Lundin et al. (2004) conducted an LCA for the comparison of end uses options for biosolids. There were some other studies that captured a wider range of environmental impacts from both treatment of sewage sludge followed by end use or disposal options (Brown et al., 2010). McDevitt et al. (2013) used the results of their LCA to encourage the engagement of the small community of Kaikoura (New Zealand) with waste management decision-making and the environment. Other studies were dedicated to the development of LCA tools for the wastewater treatment sector. The Biosolids Emissions Assessment Model (BEAM) by Brown et al. (2010) is a GHG calculator developed for the Canadian Council of Ministers of the Environment (CCME) to allow municipalities to estimate GHG emissions from biosolids processing through end use or disposal. SiSOSTAQUA by Pasqualino et al. (2009) has been used for the environmental assessment of wastewater treatment and reuse. SiSOSTAQUA includes a full range of impact categories, while BEAM is an Excel-based model and mainly focuses on global warming.

LCA is a versatile tool that extends beyond just evaluating products or processes at a process level. It has been effectively utilized for regional infrastructure planning and policy analysis. Here are some examples of LCAs that pertain to sewage sludge or biosolids management. Cartmell et al. (2006) used an LCA to evaluate the feasibility of incinerating sewage sludge and municipal solid waste from environmental, economic, and social perspectives. This study also incorporated other analytical methods such as cost-benefit analysis, risk characterization, and sustainability appraisal (Cartmell et al., 2006). Moreover, the management of sewage sludge has served as a case study for refining LCA methodologies. Poulsen and Hansen (2003) concentrated on developing methods to quantify the usage of non-renewable resources when treating biosolids. Renou and others (2008) examined how different life cycle impact assessment methodologies can influence LCA outcomes (Renou et al., 2008). Finally, Hospido et al. (2010) evaluated the impact of emerging pollutants like pharmaceuticals and personal care products on overall toxicity potential in biosolids.

2.11.2. Methodological Assumptions

The functional unit is a crucial aspect of any LCA study as it provides a reference to which all inputs and outputs can be related, enabling the comparison of different systems. The choice of functional unit can vary greatly depending on the specific goals and context of the study. A mass-based approach was the most common practice whereas volume-based functional units were used to a lesser degree (Yoshida et al., 2013). More recently, studies have started to use functional units that reflect the specific service or goods provided by treatment of sewage sludge. Hong and Li (2011) focused on the production of final products (Portland cement), while one terajoule (TJ) of steam production from sludge incineration was studied by Liu et al. (2011). In the context of biosolids land application, the agronomic value of biosolids through land application was highlighted by two studies where the provision of phosphorus (P) to plants was used as function. Linderhol et al. (2012) introduced a functional unit of “11 kg of pure P to agricultural land” whereas Pradel & Aissani (2019) used “an annual production of 1 kg of P available for plants in mineral form”. These Functional units reflect the important role of biosolids in providing essential crop nutrients for plant growth.

The system boundary defines which technical processes will be included in the study. Raw sludge is used as the starting point for the LCAs in many assessments (Heimersson et al., 2017; Houillon & Jolliet, 2005; Liu et al., 2013; Suh & Rousseaux, 2002). Other studies expanded the system boundary to include the wastewater treatment processes and accounted for the inputs and outputs of this system (Pradel & Aissani, 2019; Sablayrolles et al., 2010)

An important distinction between the LCA studies is whether sewage sludge was regarded as a product and resource or a waste. This distinction can be made by examining the functions, Functional units, the allocation rules and the system boundaries of the LCA studies and can have an impact on the LCA method and consequently, the results. Some authors such as Suh and Rousseaux (2002) described the sewage sludge treatment process as a waste reduction step with no energy recovery involved. Other authors, such as Hospido et al. (2004), Sablayrolles et al. (2010), Brown et al. (2010) and Lundin et al. (2004), have considered sludge treatment as a recovery process for nutrients such as phosphorus (K) and nitrogen (N) and energy recovery, sometimes even material recovery in the case of Houillon and Jolliet (2005). In all cases, the sludge entering the modeling system is not charged with an environmental burden associated with its production at the WWTP. In 2016, the critical review of Pradel et al. challenged the "zero burden assumption" especially when the sludge treatment is designed to produce sludge-based fertilizers. The status of sewage sludge as a "waste" was questioned and a paradigm shift from "waste" to "co-product of the WWTP" was asserted. For that reason, Pradel et al. (2019) later assessed the environmental impacts of sludge-based phosphate fertilizer production using a "product" LCA perspective instead of a "waste" LCA perspective. Consequently, upstream production of sludge was considered by allocating part of the environmental burdens of wastewater treatment to sludge production. According to Pradel et al. (2019), when considering upstream emissions, sludge-based phosphate fertilizers seemed to have a greater environmental impact than their mineral phosphate counterparts. When it came to the Life Cycle Impact Assessment, the choice of methodology is largely based on the location of the study as well as the selection of environmental impact categories (Yoshida et al., 2013). In a review done by Yoshida et. Al (2013) on 35 published studies on LCA of sewage sludge, IMPACTWORLD+ (Bulle et al., 2019) and the guideline by the Intergovernmental Panel for Climate Change (IPCC) (Myhre et al., 2013) were identified as the most favored life cycle impact assessment methods among sewage sludge LCAs. When GHG emissions were the sole focus of the study, the national emission reporting guideline set by the IPCC was also used (IPCC, 2006). By far the most common impact category considered was global warming potential in kg-CO_2e. which aligns with the increased interest in mitigating GHGs from the wastewater treatment sector as well as the agricultural sector.

2.11.3. Biosolids treatments and disposal in LCAs

Various biosolids treatments have been compared using LCA, namely thickening and dewatering, stabilization, and thermal treatments such as mono-incineration and co-incineration (Corominas et al., 2013; Yoshida et al., 2013). The most common stabilization practices are alkaline stabilization, anaerobic digestion and composting (Yoshida et al., 2013).

2.11.3.1. Alkaline Stabilization Modelling

Alkaline stabilization was often modelled by accounting for chemical and energy inputs. Suh & Rousseaux accounted for 200 kg of quicklime per 1 tDM of sludge and five kWh of electricity which is consumed for pumping the sludge and for mixing it. Houillon & Jolliet and Alanya et al. (2015) used respectively 400 kg and 130 kg of lime per tDM. There is a consensus in the literature over quicklime manufacturing contributing significant amounts of CO_2 (Alanya et al., 2015; Houillon & Jolliet, 2005; Murray et al., 2008). Houillon & Jolliet (2205) estimated significant amounts of CO_2 being emitted, around 583 kg of CO_2 per tDM. Alanya et al. (2015) concluded that most energy extraction related to alkaline stabilization originated from quicklime manufacturing. Murray et al. (2008) reported that lime addition made up 93% of overall energy consumption and 50% of nitrous oxide emissions. Although alkaline stabilization is less capital and labor-intensive than anaerobic digestion and composting, lime addition could have a great impact on the overall environmental performance because of the high embedded energy and material requirement for lime addition (Yoshida et al., 2013)

2.11.3.2. Composting Modelling

Composting was modelled by adding inputs of fuel such as diesel and electricity for material turning, screening of final products, ventilation and odour control, and outputs of emissions from energy use and fugitive gas releases (Brown et al., 2010; Yoshida et al., 2013).
Studies such as Murray et al. (2008), Poulsen & Hansel (2003) and Sablayrolles et al. (2010) also included the addition of bulking agents such as woodchips, sawdust, and yard wastes as well as the transportation of these materials to the composting facilities. Other studies did not mention the input of any sort of bulking agents (Suh & Rousseaux, 2002; Tarpani et al., 2020). Brown et al. (2010) accounted for fugitive emissions of CH_4 and N_2O during the composting process. Based on 12 field measurements, the study assumed 0-2.5% of C would be emitted as CH_4 and 0-4.6% of

input N would be emitted as N_2O. Brown et al. (2010) concluded that minimizing fugitive emissions of CH_4 and N_2O during biosolids processing due to poor composting management is key to reducing the overall life cycle global warming impact of this practice.

2.11.3.3. Anaerobic Digestion Modelling

Anaerobic digestion of biosolids is a process frequently modeled in Life Cycle Assessment (LCA) studies, as highlighted by Yoshida et al. (2013). Much like alkaline stabilization and composting, the modeling of anaerobic digestion often involves a straightforward input-output energy calculation. `The biogas generated during this process is typically utilized within the system itself, serving as a source for process heating and electricity (Brown et al., 2010; Poulsen & Hansen, 2003). The biogas generation rate is primarily calculated using either a volatile solids destruction rate or a COD removal rate (Hong, Hong, Otaki, & Jolliet, 2009; Murray et al., 2008; Suh & Rousseaux, 2002), while some studies use operational data collected at the treatment facilities (Hospido, Moreira, Fernández-Couto, & Feijoo, 2004; Poulsen & Hansen, 2003). Despite the recognized benefits of biogas substitution, not all studies, such as that by Bridle and Skrypski-Mantle (2000), accounted for this factor in their LCAs. Furthermore, few studies account for fugitive emissions during treatment. For instance, Poulsen and Hansen (2002) included CH_4 emissions for biogas plants and gas engines, with respective losses of 2% and 3%. Other studies also included CO_2, NO_2, N_2O, and particulate matters, with emission rates either collected from plant facilities or based on the literature. Consequently, studies examining the global warming potential of anaerobic digestion often document lower and even negative impacts due to biogas production and the offset of GHG emissions from the use of natural gas or fossil fuel-based electricity (as will be highlighted later in the literature). However, it's important to note that fugitive emissions such as CO_2, CO, CH_4, N_2O, and non-methane volatile organic compounds occur during this process and should be accounted for when calculating the global warming potential of anaerobic digestion. Field measurements of fugitive emissions from sewage sludge treatment processes are limited and when included, they are largely based on the assumption provided by other GHG accounting guidelines (i.e., IPCC guideline).

2.11.4. Land Application of Biosolids

Modeling land application of biosolids in LCA studies can be complex as it varies significantly based on the specific goals and scope of the study and is impacted by many factors. Generally, this process is modelled by accounting for direct and indirect emissions associated to this practice. Direct emissions associated with the land application of biosolids are emissions that occur directly from the biosolids themselves and include emissions of GHGs and ammonia (NH_3). The decomposition of organic matter in the biosolids can release GHGs such as CO_2, CH_4, and N_2O. These emissions to air are often calculated using emissions factors from Ecoinvent, the IPCC national GHG inventory or from other literature (Yoshida et al., 2013). Brown et al. (2010) excluded the direct emissions of CH_4 from the soil in their study, citing their minimal contribution to overall emissions. However, they did include them in the modelling of storage of biosolids prior to land application. The aforementioned study quantified N_2O emissions from soils, attributing them to 1% of the total nitrogen added. This calculation was based on the default IPCC factor for N_2O emissions from fertilizers, compost, and biosolids. While Brown et al. excluded CH_4 from biosolids-amended soils, Houillon and Jolliet (2005) accounted for CH_4 during storage and after spreading citing that these CH_4 emissions are significant in the global warming balance. The study also considered biogenic CO_2 emissions due to the degradation of organic matter from different point sources including the land. However, as biogenic CO_2 does not increase global warming, it was deducted from the balance. While some studies such as that of Hospido et al. (2010), Lundin et al. (2004) and Sablayrolles et al. (2010) accounted for direct emissions from the land application of biosolids, the sources of these emissions were not clearly mentioned or documented. Sablayrolles et al. (2010) included an extensive list of air emissions from sewage sludge application, but they were partly from operation of heavy equipment in the field and thus the exact contribution to global warming from the land application is not determined. No study has been identified that incorporates experimental measurements of emissions from biosolids-amended soils into their assessments. Some studies did not even consider emissions to air as a parameter to include to model biosolids used on land (Bridle & Skrypski-Mantele, 2000; Murray et al., 2008; Peters & Rowley, 2009). In the context of modeling land application in LCA studies, Yoshida et al. (2013) advocate for a more consolidated approach. They recommend quantifying fugitive emissions as a means to enhance the integrity and robustness of the assessment.

Indirect emissions that have been accounted for in the modelling of land-application of biosolids included emissions linked to the transportation of biosolids to the field, operation of the agricultural machinery for spreading and incorporating the biosolids into the land. Transportation and agricultural machinery were modelled using diesel quantity used for operations and distances travelled (Brown et al., 2010). Transport of biosolids to the field was found to be a significant source of GHG emissions when distances and quantities transported are large ((Johansson et al., 2008; Suh & Rousseaux, 2002). In addition, Alvarez-Gaitan et al. (2016), stated that biosolids water content is a key driver for transport emissions and directly linked high biosolids water content to a greater number of trucks to move the wet cake from the plant to its destination, insinuating that it is critical to minimize GHG emissions associated to transport to biosolids land application sites.

In addition, when modeling the land application of biosolids, the displacement of synthetic fertilizer was almost always taken into account across the literature as studies assumed that the nutrients (N,P, & K) in the sludge could be substituted for use of conventional fertilizer (Yoshida et al., 2013). This substitution was carried through a system expansion where the process of production, transport and spreading of synthetic fertilizer were avoided and given a negative value. Lundin et al. (2004) and Johansson et al. (2008) have demonstrated that a system expansion for avoided fertilizer is significant. The saved N_2O emissions from the utilization of mineral fertilizer and avoided CO_2 from the manufacture of mineral fertilizer contribute to reduction in GHG emissions (Yoshida et al., 2018). Houillon & Jolliet (2005) highlighted the importance of the substitution and showed that turning sludge into a resource constitutes an efficient way to compensate for treatment emissions.

Lastly, an emerging concern related to the agricultural land application of biosolids is the potential introduction of contaminants into arable lands. A prevalent argument in the literature suggests that the use of biosolids on land could result in the accumulation of heavy metals and other pollutants in soils (Corominas et al., 2013; Yoshida et al., 2013). This presents a clear paradox: the nutrient recovery value of biosolids land application is undeniable, yet the associated health risks spark public debate.

Given the conflicting aspects of this practice, studies such as those by Sablayrolles et al. (2010) and Hospido et al. (2010) have particularly emphasized human toxicity via plant ingestion. Numerous studies have incorporated considerations of pathogens, heavy metals, and organic compounds originating from pharmaceutical and personal care products into their assessments (Hospido et al., 2010; Hospido, Moreira, Martin, Rigola, & Feijoo, 2005; Johansson et al., 2008; Sablayrolles et al., 2010; Suh & Rousseaux, 2002). LCA studies quantifying the environmental impact of heavy metals applied to soil have acknowledged high levels of uncertainty. This uncertainty arises from the unknown factors affecting the behavior of heavy metals in soil after land application. Moreover, significant uncertainties and limitations exist in the impact assessment models, particularly those used to calculate the toxicity impact category relevant to heavy metal emissions (Alanya et al., 2015; Hospido et al., 2005; Peters & Rowley, 2009). In conclusion, while many LCA studies have highlighted the valuable benefits of applying biosolids to land, such as nutrient recovery and the displacement of synthetic fertilizer usage, there is a clear need for further research. Specifically, more accurate factors for the discharge of heavy metals into the soil need to be determined to provide a more comprehensive and realistic assessment of the environmental impacts to human and ecological health (Yoshida et al., 2013).

2.11.5. Comparative LCA studies in the Literature

The LCA framework was extensively applied in the field of sewage sludge management (Corominas et al., 2013; Yoshida et al., 2013). The primary objective across LCA studies in the literature is to compare various combinations of biosolids treatment and disposal techniques, with the aim of assessing their impacts across diverse environmental categories. Interestingly, while each study examined different biosolids treatment and disposal techniques, the study by Suh & Rousseaux (2002) stands out. It was the only study that considered comparing the scenarios of anaerobic digestion (anaerobic digestion), alkaline stabilization (ATB), and composting (COM) of biosolids followed by land application. Despite the multitude of scenario combinations assessed and compared in the literature, it's crucial to understand that the results of these studies are not directly comparable. This is due to the inherent specificity of LCA, a tool that has proven to be invaluable in this context. The results of an LCA are intrinsically linked to the goal and scope of each individual study, thereby reflecting the unique local conditions under investigation. For example, the geographical area and other local conditions affected the GWP for each study (Yoshida et al., 2013). Brown et al. (2011) calculated the GHG emissions associated with anaerobic digestion followed by land application and found that the results ranged from -26 to 43 kg-CO_2e/t of dry sludge. This difference was largely due to variations in background emissions of electricity production: 0.733 kg-CO_2e./kWh electricity generation in the Canadian province of Nova Scotia, and 0.01 kg-CO_2e/kWh electricity in Quebec. Secondly, variations exist in process and emissions data as the assumptions made for energy and chemical consumptions vary greatly between LCA studies (Brown et al., 2010). Despite LCA studies not being comparable, this literature review reveals common conclusions. Several studies that modelled anaerobic digestion as one of the biosolids treatment techniques found that it had a lower impact on global warming and were the best-performing treatment methods over other techniques (Brown et al., 2010; Corominas et al., 2013; Lee et al., 2020; Suh & Rousseaux, 2002; Tarpani et al., 2020; Teoh & Li, 2020; Zhuang et al., 2022). Brown et al. (2010) concluded that the programs that had anaerobic digestion followed by land application resulted in the lowest emissions (-26 and -23 Mg CO_2e 100 Mg-1 biosolids (dry wt.). Similarly, Suh & Rousseaux (2002) concluded that the combination of anaerobic digestion and agricultural land application was the most environmentally friendly tanks to less emissions and less consumption of energy. As Tarpani et al. (2020) also established, anaerobic digestion with recovery of nutrients and electricity had the lowest environmental impact, particularly regarding global warming. A recent Canadian study by Zhuang et al. (2022) concluded that anaerobic digestion coupled with agricultural land application have an expected global warming impact at least 60% lower than the alternative treatment methods studied. Conversely, McDevitt et al. (2013) found that composting sewage sludge resulted in a higher carbon footprint compared to landfilling. However, this study's comparison of a treatment technique (composting) with disposal techniques (land application and landfilling) did not take into account the nutrient value of composting or the displacement of fertilizer when this compost is applied to land. Instead, the study modeled the scenario where composted sewage sludge is sold to the community, contrasting it with the scenario of pastoral land application where sewage sludge is directly applied to the land. Lastly, Murray et al. (2008) discovered that anaerobic digestion (anaerobic digestion) of sewage sludge resulted in negative emissions, as the biogas produced was utilized for electricity generation, thereby replacing electricity from a coal-fired power plant and avoiding the use of fossil fuels. However, the same study highlighted a critical finding regarding the alkaline stabilization (ATB) of sludge. The production of lime, a significant component of this process, was found to have a substantial impact on climate change, contributing to 93% of the total fuel consumption. This made ATB the least preferable treatment option in the study due to its high environmental impact.

Houillon and Jolliet (2005) and Lundin et al. (2004) are two studies that only assessed the GWP of disposal options without looking at the impact that biosolids treatment would have on the overall LCA global warming impact. The latter found that land application of sewage sludge was the worst disposal option following landfilling and the second concluded that it was land application that was the least preferable option. Both studies compared land application to other disposal and end-use options such as incineration, wet oxidization pyrolysis of dried sludge.

2.12. Research objectives

This literature review has underscored the extensive research focused on assessing the environmental impacts of various treatment and end-use options for biosolids. However, a holistic assessment of biosolids management, from production to land application, is notably lacking in the existing literature. Only one study, conducted by Suh & Rousseaux (2002), included anaerobic digestion (anaerobic digestion), alkaline treated biosolids (ATB), and composting (COM) - the most practiced treatment techniques in Canada - followed by agricultural land application. Yet, this study relied heavily on literature data and default values for estimating GHG emissions from land application. Given the sensitivity of LCA results to geographical locations, there is a clear need for studies that reflect the specific climate and soil conditions in Canada. Furthermore, Yoshida et al. (2013) stated that emissions from the land application of biosolids represent the largest source of uncertainty in LCAs. Therefore, more accurate and geographically specific estimations are needed to refine the modeling of biosolids land application, filling a critical gap in the current body of research.

The existing literature reveals a notable gap in understanding the GHG emissions associated with various biosolids management options in Quebec, Canada. This further highlights the pressing need to quantify and compare these emissions to identify the most environmentally favorable biosolids management practice based on global warming potential.

The central research question guiding this study is:

Which biosolids treatment and application technique would best limit GHG emissions from the agricultural sector in Canadian climate and soil conditions?

The main objectives of this study are:

1- To design a comparative carbon footprint assessment of different scenarios of biosolids production technologies and application methods
2- To quantify GHG emissions associated with each scenario.
3- To determine the Global Warming Potential (GWP)
4- To provide recommendations for farmers, WWTPs operators and climate policy makers on how to limit agricultural emissions based on our results.

Chapter 3 Methodology

3.1. Goal of the Study

The aim of this study is to quantitatively assess the GWP of various treatment and disposal pathways for municipal wastewater biosolids in Quebec, Canada. The objective is to identify the most efficient system for biosolids management by comparing the GWP of 15 different management scenarios. This assessment is carried out using the Life Cycle Assessment (LCA) approach as stipulated by ISO 14040-44. The software used for the modelling of the life cycle inventory and life cycle impact assessment is OpenLCA (v.1.11. GreenDelta, Berlin, Germany, www.openlca.com). OpenLCA is an open-source software for LCA and sustainability assessment, created and maintained since 2006 by GreenDelta, Berlin. The modelling was done using primary data and completed by using secondary data from Ecoinvent 3.6 Cut-Off (2019) (Wernet et al., 2019), a life cycle inventory database. The case study is based on the Montreal experimental study conducted by McGill University as part of the Biosolids project. Although an LCA methodology was adopted, the assessment is referred to as a Carbon Footprint as the primary focus of this thesis is on the GWP impact of the scenarios. Because deliverables of this project to AAFC were already set, the focus of this study is to assess the global warming potential of the application of biosolids to the land. The main objective of this study is to assess and quantify the GHG emissions associated with the practice of biosolids application to Canadian land as required by AAFC. LCA is only a tool that answers the objectives of the study and is thus not the center of it.

3.2. Montreal Experimental Design

The field experiment had three treatment factors: fertilizer type, rate of biosolid application, and method of application. Silage corn was grown with either commercial urea, digested biosolids, alkaline-stabilized biosolids, or composted biosolids. Biosolids are often used in combination with mineral fertilizer, so each biosolids type was applied either at full-rate or in combination (1:1 ratio) with urea. The fertilizers were either surface-spread or incorporated by cultivation, to test the effect of incorporation on GHG emissions. An unfertilized treatment or "zero-fertilizer" treatment served as the negative control. In total, there were 15 treatments organized. Each experimental plot received 39 kg-N/ha in the form of calcium ammonium nitrate as a starter dose of N at seeding. The remaining N was applied to the fertilized treatments as either of the biosolids before seeding and/or commercial urea applied at the six-leaf stage, depending on the treatment, to achieve a targeted 120 kg of applied available N per hectare, annually. For the biosolids treatments without urea, the target N was achieved by adding a total of 162 kg-N/ha as biosolids, with 50% assumed to be available to the crop, i.e., 81 kg available N/ha from biosolids and 39 kg-N/ha from the starter fertilizer. Then for the treatments receiving half N from biosolids, the target was achieved by applying 39 kg-N/ha as starter fertilizer, 40.5 kg-N/ha as urea, and 40.5 kg available N/ha as biosolids (162 kg total N as biosolids but assuming only 50% was available to the crop) (Obi-Njoku et al., 2023; Obi-Njoku et al., 2022).

The three technologies for treating biosolids were:

- Anaerobic digestion
- Alkaline stabilization
- Composting

Types of application:

- Surface-spread
- Incorporated

Urea make-up:

- Without urea (all additional fertilizer after seeding was applied as biosolids)
- With 50% urea

Controls:

- Urea surface-spread
- Urea incorporated
- No fertilizer

3.3. Scope definition of the Study

3.3.1. Function and Functional Unit

To overcome the differences in performance characteristics of the scenarios studied, an LCA or carbon footprint makes the comparison on the basis of the function or functions fulfilled by these systems. In the present case, the systems studied are treated municipal wastewater sewage sludge or biosolids. The function they fulfill is to “grow corn silage cropping”. The functional unit (functional unit) to which the inventory calculations and the carbon footprint assessment relate, then quantifies this function and places it in its geographical context. In the case of this study, the 15 different scenarios of biosolids treatment and land-application need to be compared on the same comparative bases. The recommended application rate of nitrogen fertilizer for silage corn crops in Quebec may vary depending on factors such as soil type, previous crop, and weather conditions. However, according to the Quebec Reference Center for Agriculture and Agri-food (CRAAQ), a general guideline for nitrogen application for silage corn in Quebec is to apply 120-170 kg of nitrogen per hectare for optimal yield and quality (CRAAQ, 2003). A functional unit was therefore defined as: "To provide 120 kg of N nutrient per 1ha for corn silage cropping in Quebec between 2017-2019."

3.3.2. System Boundary

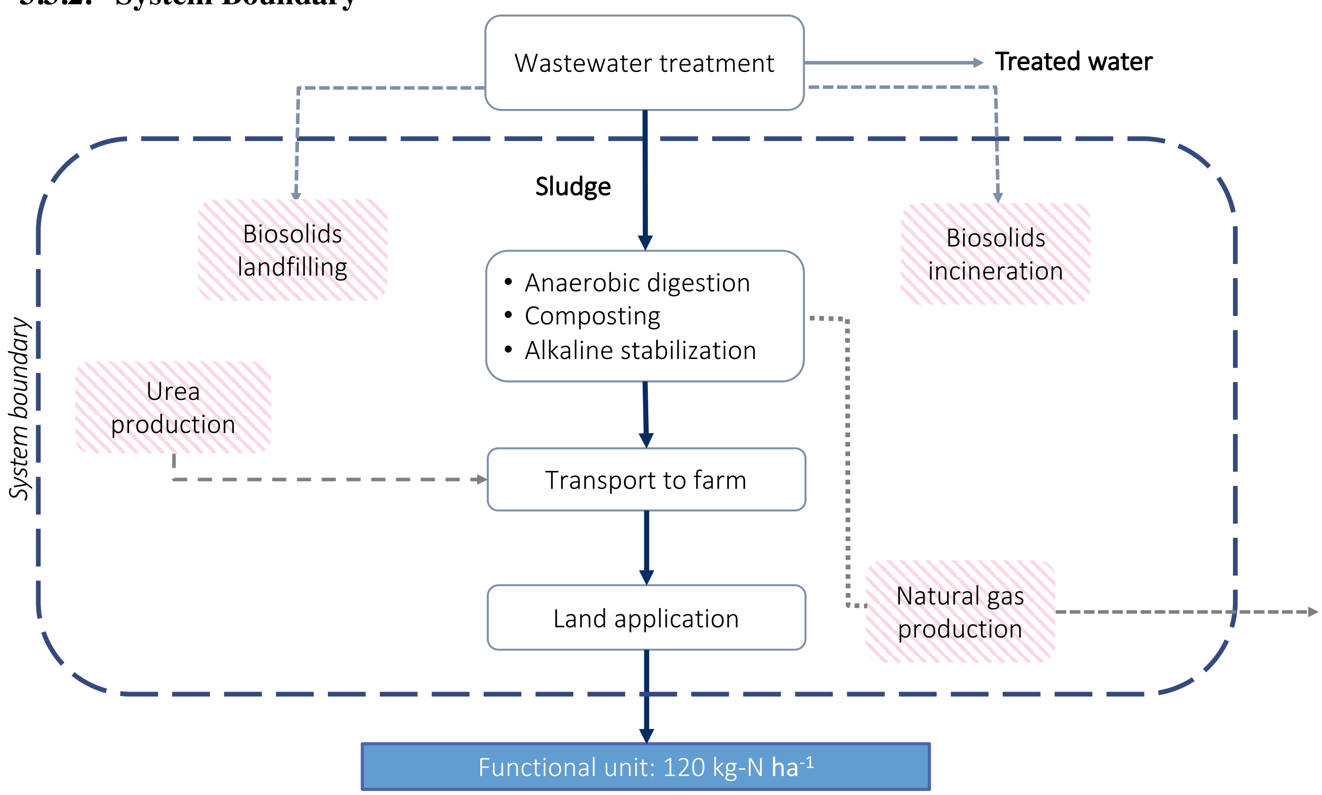

Figure 3.1. System boundary of study

The system boundary was designed as such that it fulfilled the functional unit of the study. As can be seen in figure 3.1, the system starts at the point where sewage sludge is transported to the wastewater treatment facility where it is subjected to either anaerobic digestion, alkaline stabilization, or composting treatment. Then, the treated sewage sludge or biosolids are transported to a farm where they will be land-applied on agricultural soils. The system boundary was expanded to include urea production, transportation, and land application for comparison purposes.

As discussed in section 2.5.2.1, multifunctionality of a system can be solved either by dividing the unit processes to be allocated in two or more sub-processes or by expanding the system to include the additional functions related to the co-products. In this case, system expansion has been opted as a method to deal with the multifunctionality of the system under study. Hence the system has been expanded to model avoided GHG emissions from processes that have been avoided due to the practice of treated sewage sludge and land-application. The system has been expanded to include avoided processes of producing natural gas due to the co-production of biogas during anaerobic digestion. Similarly, scenarios of landfilling and incineration of sewage sludge have also been considered in the case where these disposal options were opted versus the practice of biosolids land-application. All in all, and based on the Montreal experimental design, 15 different scenarios are analysed in this study with two additional scenarios showcasing alternative disposal options. The GWP of these scenarios would be analysed and compared to determine which of them is the most favourable vis-à-vis avoided environmental impacts.

The 15 scenarios are listed as follows:

Table 3.1. Fifteen experimental treatments compared in this study

Treatments	
Control zero	No fertilizer applied on the soil except for the stater fertilizer.
UF_SS	Full-rate urea fertilizer application, surface-spread on the soil
UF_ INC	Full-rate urea fertilizer application, incorporated into the soil.
AD_SS	Full-rate application of digested biosolids, surface-spread on the soil
AD_INC	Full-rate application of digested biosolids, incorporated into the soil.
ATB_SS	Full-rate application of alkaline-stabilized biosolids, surface-spread on the soil.
ATB_INC	Full-rate application of alkaline-stabilized biosolids, incorporated into the soil.
COM_SS	Full-rate application of composted biosolids, surface-spread on the soil.
COM_INC	Full-rate application of composted biosolids, incorporated into the soil.
UF+AD_SS	digestion SS: Half and half blend of urea and digested biosolids, surface-spread on the soil
UF+AD _INC	Half and half blend of urea and digested biosolids, incorporated into the soil.
UF+ATB_SS	Half and half blend of urea and alkaline-stabilized biosolids, surface-spread on the soil
UF+ATB_INC	Half and half blend of urea and alkaline-stabilized biosolids, incorporated into the soil
UF+COM_SS	Half and half blend of urea and composted biosolids, surface-spread on the soil
UF+COM_INC	Half and half blend of urea and composted biosolids, incorporated into the soil.

3.3.3. Data Requirements

The data utilized in this project was sourced from various locations. Primary data was gathered from wastewater sewage sludge treatment facilities in Canada, using interviews and questionnaires. The biosolids used for the Montreal experiment were supplied by the following facilities: Digested biosolids were provided by the Centre de Valorisation des Matières Organiques (CVMO), Saint-Hyacinthe, Quebec; alkaline-stabilized biosolids were sourced from Walker Industries (N-Viro Systems), Halifax, Nova Scotia and composted biosolids were obtained from Gaudreau Environment Inc., Victoriaville, Quebec.

To gather operational data specific to the biosolids treatment techniques at each facility, an Excel spreadsheet was created. This spreadsheet included a table for operators at each facility to fill out, detailing various characteristics of the biosolids, as well as the inputs and outputs required for treatment. Unfortunately, only one of the three tables was completed, specifically the one sent to Walker Industries for alkaline-stabilized biosolids. The three tables are included in the appendix 1,2 and 3. In addition to the primary data, relevant information was also extracted from existing literature on related LCA studies. In instances where precise data was unavailable, expert estimations were employed to fill the gaps.

Finally, an important and unique contribution of this study to the existing literature is the use of experimental data to conduct the modelling of land application. Field measurements for the Montreal experiment include GHG emissions, in particular CH_4, CO_2 and N_2O, total nitrogen concentration in the aboveground biomass of the maize crop (mg/g) and silage yield (dry t/ha). Measured CO_2 and N_2O emissions in 2018, out of the three-year (2017 – 2019) data, was selected to conduct the carbon footprint assessment.

3.3.4. Data Quality Assessment

The accuracy of the results and conclusions of life cycle modeling is closely tied to the quality of the inventory data. Ensuring this data aligns with the study's objectives is crucial. While ISO doesn't provide a specific method for data quality assessment, this study adopted a five-criteria approach: reliability, completeness, temporal correlation, geographical correlation, and technological correlation. Each data point in the inventory is rated on a scale from 1 (highest quality) to 5 (lowest quality) based on these criteria.

Table 3.2 outlines the criteria for data validation. These standards relate to the data's relatability and representativeness. It's worth mentioning that this evaluation is streamlined to keep the life cycle impact assessment process efficient, yet it offers a comprehensive overview of the type of inventory collected.

Table 3.2. Criteria for qualifying the reliability and representativeness of the data

Score	Criteria for qualifying data reliability (quantities)
1	Verified data measured or calculated in the field - This data fulfills the “reliability/accuracy” criterion required for the case under study
2	Verified data, partly derived from assumptions or Unverified data derived from measurements (documents provided by the agent or literature) – this data is deemed sufficiently precise/reliable by the team of analysts for the case under study
3	Unverified data, partly based on assumptions or Quality estimate (made by an expert) – this data is deemed usable by the analyst team, but its reliability/accuracy could be improved
4	Roughly estimated data - This data does not meet the “reliability/accuracy” criterion required for the case under study
Score	**Criteria for qualifying the representativeness of the data (process)**
1	Field data (from the framework under study), laboratory data - This data fulfills the "representativeness" criterion required for the case under study
2	Good geographical or technological representativeness of the selected process – this data is deemed sufficiently representative by the team of analysts for the case under study
3	Data relating to the same process or material, but referring to a different technology (e.g., representative process available in the Ecoinvent database) – This data is considered usable by the team of analysts, but its representativeness could be improved
4	Inadequate geographical or technological representativeness. The data sought is not easily accessible, use of another process as an approximation - This data does not meet the "representativeness" criterion required for the case under study

The "reliability" aspect of data quality pertains to the accurate measurement of flows, including material and energy, transportation distances, and release amounts. On the other hand, "representativeness" in data quality speaks to the geographical and technological accuracy and comprehensiveness of the chosen generic data modules (or processes). Lastly, the potential impact contribution indicates the effect of the evaluated process or parameter on the outcomes, based on its average contribution to the studied impact categories. For ease of interpretation, a color code has been incorporated, as detailed in the table 3.3.

Table 3.3. Color code for process contribution

Contribution		Quality	
0–5%	Potentially small or negligible contribution	1	Fulfills the criterion for the case under consideration. Ex. Data validated by an expert in the field.
6–10%	Potentially influential contribution	2	Considered sufficiently representative. Ex. Generic data, specific or applicable to the Quebec context
11–50%	Strong potential contribution	3	Considered usable but could be improved. Ex. Generic data implying fairly close substitute data.
51–100%	Very strong potential contribution	4	Does not meet the criterion for the case under consideration. Ex. Substitute data; rough estimate.

Typically, a score of "1" indicates an excellent assessment, whereas a score of "4" suggests that the data should be improved in order to meet the various quality criteria. Thus, the processes for which the quality of the data is considered to be limited or insufficient are highlighted in red (score "4") and the processes that can be improved are in orange (score "3").

With respect to contribution, a range of values is presented. It indicates the minimum and maximum contribution of the assessed process according to Climate Change. The overall contribution of the evaluated process (color of the box) was established according to its maximum contribution, all indicators combined. In parallel with the evaluation of data quality, an estimate of the contribution of processes (i.e., to what extent the modeled process contributes to the overall impact score of the studied system) was carried out in Appendix 7.4, 7.5 and 7.6. Low-quality data may be appropriate in the case of a process whose contribution is minimal. Conversely, high-quality data should be collected for processes that have a significant influence on the conclusions of the study.

3.3.5. Impact Assessment Method

The impact assessment method used to translate the inventory results of the scenarios into potential contributions to various impacts is the IPCC method for a 100-years horizon of global warming potential (GWP100) (Myhre et al., 2013). This study does not progress to the normalisation phase; it ends at the characterisation step. Impact assessment includes only the assessment of the global warming impact.

3.3.6. General Assumptions of Study

The following assumptions were made in the study:

- All biosolids treatment facilities are assumed to be in Quebec, Canada. For instance, the electricity consumed for alkaline biosolids, which is produced in Halifax, is assumed to be sourced from the Quebec electrical grid.
- It is assumed that all sewage sludge is treated at the same facility and applied at the same farm. This simplification allows for a more streamlined analysis, although it may not fully capture the variability in practices across different facilities and farms.
- The biogas generated during the anaerobic digestion process is assumed to be returned back into the system. This assumption is based on the common practice of using biogas as an energy source within the treatment facility, thereby reducing the need for external energy inputs.
- In the experimental design, nutrient application was calculated on the basis of crop requirements, and any nutrients remaining in the soil from previous growing seasons were not considered. This assumption simplifies the nutrient accounting process, but it may not fully capture the long-term nutrient dynamics in the soil.

3.3.7. Limitations of Study

- This study primarily focused on the GWP impact, aligning with the main objective of the project. Initially, acidification and eutrophication potentials were considered for inclusion in the study. However, due to the lack of necessary data for accurate modeling (e.g., nitrate and phosphate leaching to groundwater), these aspects were ultimately omitted from the analysis.

- Carbon sequestration, a significant factor in the carbon cycle and climate change mitigation, was not included in the analysis due to data limitations.
- Similarly, the analysis of heavy metals was excluded because a consistent mass balance could not be reliably assessed. Consequently, the impacts on ecotoxicity and human toxicity will not be interpreted in the LCA.
- While this study does not include an uncertainty or sensitivity analysis, a data quality assessment of the inventory was conducted to ensure the reliability and accuracy of the data used in the study. This assessment helped to identify any potential limitations or biases in the data, thereby enhancing the robustness of the study's findings.

3.4. Life Cycle Inventory

3.4.1. Biosolids Characteristics and Application Rate

Based on the biosolids properties collected, the Total Nitrogen content was calculated for each biosolids type.

Table 3.4. Biosolids total nitrogen content

Treatment	TN[1] (kg-N/kg biosolids dry basis)	Dry matter (%)	Source[3]
Digested biosolids	0.0559	20%	CVMO[2], Saint-Hyacinthe, QC
Alkaline-stabilized biosolids	0.01	62%	N-Viro Systems, Halifax, NS
Composted biosolids	0.00475	38%	Gaudreau Environnement Inc., Victoriaville, QC

[1] Total nitrogen
[2] Centre de référence en agriculture et agroalimentaire du Québec
[3] Unpublished data shared during personal communication with the companies.

The amounts of biosolids required to be applied on the land to achieve full-rate and half-rate application were then calculated based on the total nitrogen content of each biosolids type in order to meet the functional unit of 120 kg-N/ha for corn silage cropping. Consequently, the quantities presented in Table 3.5 correspond to the amount of each type of biosolids required for application.

Table 3.5. Biosolids application rate

Treatment	kg of biosolids required for full-rate application of 81 kg-N/ha	kg of biosolids required for half-rate application of 40.5 kg-N/ha
Digested biosolids	14,276	7,138
Alkaline-stabilized biosolids	26,129	13,065
Composted biosolids	89,281	44,640

3.4.2. Life Cycle Inventory for Biosolids Treatment

3.4.2.1. Transportation of Sewage Sludge from WWTP to Treatment Facilities

Sewage sludge is assumed to be treated at the same distance from the WWTP for all biosolids treatment techniques to simplify the comparison. Transportation distance from WWTP to the biosolids treatment facility is assumed to be 10 km. Transportation distance for organic matter to the anaerobic treatment facility is also assumed to be 10 km. Transportation is carried out by diesel trucks with a carrying capacity 7.5-16 tons (Poulsen & Hansen 2003). The transportation process is modelled in OpenLCA as *“market for transport, freight, lorry 7.5-16 metric ton, EURO5 | transport, freight, lorry 7.5-16 metric ton, EURO5 | Cutoff, S”* (Ecoinvent 2019).

3.4.2.2. Anaerobic Digestion

The data used to conduct the life cycle inventory for anaerobic digestion in this study was taken from personal communication with CVMO in the city of Saint-Hyacinthe (Guy Nadeau, Usine d’epuration, Saint-Hyacinthe, QC, personal communication, 2020). Additionally, calculations were performed using the anaerobic digestion waste energy balance tool by Valorgas

(Valorgas, version W7f-waste, University of Southampton, Southampton, UK), a spreadsheet-based anaerobic digestion modeling tool that is developed by the University of Southampton. It calculates digester size and energy requirements from specifies input feedstock materials.

To explain the operational process of anaerobic digestion, a simplified flow diagram was presented in figure 3.2. In this system, approximately 13 million m^3 of raw sewage sludge, a by-product of wastewater treatment, undergo anaerobic digestion. This process involves mixing the sludge with organic matter to facilitate digestion. The biogas generated during this treatment is subsequently purified and converted into energy, which is then reused by the plant. Notably, the system also accounts for the offset production of natural gas as the heat requirements for the digesters are supplemented by the produced biogas. Following the digestion process, the resulting digestate is transported to a nearby field for land application, thereby completing the cycle.

To begin, the modelling process took into account the inputs related to the infrastructure and construction necessary for operating an anaerobic digestion (anaerobic digestion) system. The following components were incorporated into the anaerobic digestion process:

- Digestate storehouse specific to anaerobic digestion
- Digester designed for anaerobic digestion
- Gas holder for anaerobic digestion

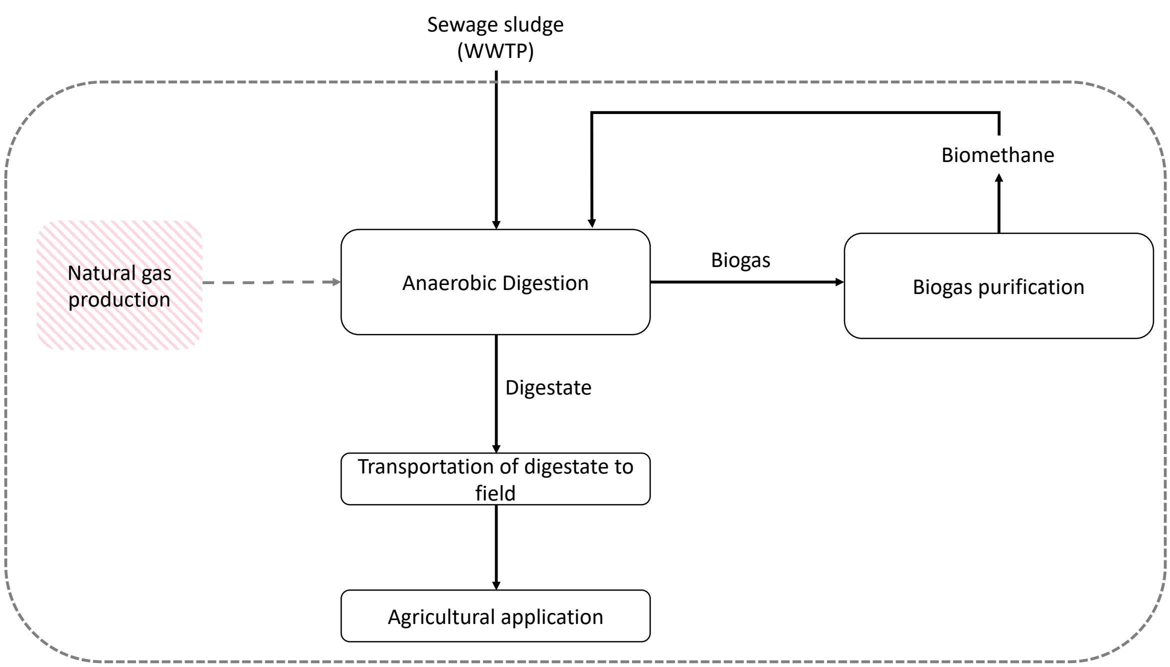

Figure 3.2. Simplified flow diagram for anaerobic digestion process (Avoided production of natural gas is here presented by the red hatched box)

The model assumed a lifespan of 30 years for the infrastructure with capacities similar to that of the digester. Each flow was input with a value of 0.02812 which represents the fraction of the infrastructure used to conduct the process for the given quantity of sewage sludge processed. The calculation used, along with the involved parameters, is detailed below:

Fraction of infrastructure used = (Sludge input anaerobic digestion + organic matter)/ (lifetime infrastructure × (digesters capacity)

Table 3.6. Parameters for infrastructure modelling in anaerobic digestion

Parameter	Unit	Value	Source
Sludge input, anaerobic digestion	kg/year	13,043,000	(Ville de Saint-Hyacinthe, 2023)
Organic matter	kg/year	161,459,000	(Ville de Saint-Hyacinthe, 2023)
Lifetime infrastructure	years	30	Default value (expert estimate)
Digester capacity	kg/year	206,850,000	(Ville de Saint-Hyacinthe, 2023)

* Centre de référence en agriculture et agroalimentaire du Québec

Electricity and heat are needed for operating and heating the digesters. The anaerobic digestion waste energy balance tool (version W7f- waste, University of Southampton, Southampton, UK) developed by Valorgas was used to obtain typical energy use requirements for the digester capacity studied. Feedstock information such as sewage sludge input, number of digesters and feedstock composition were input into the spreadsheet. The parameters used for energy usage of the digestion process as well as the flows input into OpenLCA are displayed respectively in the tables 3.7 and 3.8 below:

Table 3.7. Parameters for energy usage modelling in anaerobic digestion

Parameter	Unit	Value	Source
Electricity consumption	kWh/kg of sludge	0.061	Valorgas*
Heat consumption	MJ/kg of sludge	0.266	Valorgas

* Valorgas, version W7f-waste, University of Southampton, Southampton, UK

Table 3.8 presents the flows input into OpenLCA, that have been calculated using the parameters presented above in table 3.7.

Table 3.8. Flows for annual energy usage in anaerobic digestion modelling

Flow	Unit	Value	Ecoinvent flow name*
Electricity consumption	kWh	806,188	*market for electricity, medium voltage | electricity, medium voltage | Cutoff, S*
Heat consumption	MJ	3,480,650	*heat production, natural gas, at boiler modulating >100kW | heat, district or industrial, natural gas | Cutoff, S*

* Ecoinvent flow names are presented in this table as they are found in the database to facilitate the replication of this modelling. The name includes the process name, the system model chosen (cut-off) and the process type (system process).

The outputs considered from the anaerobic digestion process include fugitive emissions of non-fossil CO_2 and CH_4, as well as the production of biogas.

CO_2 emissions in anaerobic digestion represent fugitive emissions from biogas losses as well as CO_2 emissions from biogas flaring. To calculate total CO_2 emissions, Eq. 1 was used:

$$\text{Fugitive } CO_2 \text{ emissions} = CO_2 \text{ ratio in Biogas} \times \%\text{biogas loss} \times \text{Biogas produced} \times CO_2 \text{ density} + CO_2 \text{ emission flaring} + CO_2 \text{ emission heat production} \qquad \text{Eq. 1}$$

The parameters collected to undertake this calculation are presented in the tables below:

Table 3.9. Parameters for fugitive CO_2 emissions modelling in anaerobic digestion

Parameter	Unit	Value	Source
CO_2 ratio in biogas	%	39	(Brown et al., 2010)
Biogas loss	%	2	(Brown et al., 2010)
Biogas produced	m^3	9,381747	Valorgas[1]
CO_2 density	kg/ m^3	1.836	(National Center for Biotechnology Information, 2023a)
Biogas flared	m^3	327,245	CVMO[2]
Biogas flared	m^3	810,956	CVMO
CH_4 ration in biogas	%	61	(Brown et al., 2010)
CH_4 density	kg/ m^3	0.554	(National Center for Biotechnology Information, 2023b)
CH_4 to CO_2 ratio	kg of CO_2/ kg of CH_4	2.75	(Brown et al., 2010)

[1] Valorgas, version W7f-waste, University of Southampton, Southampton, UK
[2] Centre de référence en agriculture et agroalimentaire du Québec (Ville de Saint-Hyacinthe, 2023)

The following table 3.10, displays dependent parameters which have been calculated using the parameters presented above in table 3.9. Dependent parameters implied that these values depend on the input parameters above.

Table 3.10. Dependent parameters for fugitive CO2 emissions modelling in anaerobic digestion

Dependent parameter	Unit	Formula	Value
Net biogas	m^3	Biogas produced × (1-Biogas loss)	9,194,112
CO_2 emission flaring	m^3	(CO_2 ratio in Biogas × CO_2 density + CH_4 ratio in biogas × CH_4density × CH_4 to CO_2)	538,441
CO_2 emission heat production	m^3	Biogas heat × (CO_2 ratio in Biogas × CO_2 density + CH_4 ratio in Biogas × CH_4 density × CH_4 to CO_2)	1,334,327

CH_4 emissions in anaerobic digestion represent fugitive emissions from biogas losses. To calculate total CH_4 emissions Eq. 2 was used.

$$\textit{Fugitive } CH_4 \textit{ emissions} = CH_4 \textit{ ratio in biogas} \times \textit{biogas loss} \times \textit{biogas produced} \times CH_4 \textit{ density} \qquad \textit{Eq. 2}$$

Table 3.11. Parameters for output emissions modelling in anaerobic digestion

Parameter	Unit	Value	Source
Biogas loss	%	2	(Brown et al., 2010)
Biogas produced	m^3	9,381,747	Valorgas*
CH_4 ratio in biogas	%	61	(Brown et al., 2010)
CH_4 density	kg/ m^3	0.554	(National Center for Biotechnology Information, 2023b)

* Valorgas, version W7f-waste, University of Southampton, Southampton, UK

The parameters collected to undertake this calculation are presented in tables 3.11 and 3.12.

Table 3.12. Flows for output emissions modelling in anaerobic digestion

Flow	Unit	Value	Ecoinvent flow name
Net biogas	m^3	9,194,112	-
CO_2, non-fossil	kg	2,027,270	*carbon dioxide, non-fossil*
CH_4, non-fossil	kg	72,920.8	*Methane, non-fossil*

The biogas produced undergoes a purification process, resulting in a refined gas that can be utilized to power the digesters. To model this process, the *"biogas purification to methane 96 vol-% | methane, 96% by volume | Cutoff, U/ QC"* process in Ecoinvent 3.6 was used to model biogas purification. Infrastructure *("chemical factory construction, organics | chemical factory, organics | Cutoff, U")* and electricity *("market for electricity, medium voltage | electricity, medium voltage | Cutoff, U")* were input to operate the purification process. The following parameters were used to model the output of this process, which include purified biogas (biomethane), CO_2 and CH_4 non-fossil fugitive emissions:

Table 3.13. Parameters for the modeling of biogas purification

Parameter	Unit	Value	Source
Biogas flared	m^3	327,245	CVMO[1]
Biogas heat	m^3	810,956	CVMO
Biogas produced	m^3	9,194,112	CVMO
CH_4 density	kg/ m^3	0.554	(National Center for Biotechnology Information, 2023b)
CH_4 loss	%	4	(Brown et al., 2010)
CH_4 ratio in biogas	%	61	(Brown et al., 2010)
CO_2 density	kg/ m^3	1.836	(National Center for Biotechnology Information, 2023a)
CO_2 ratio biogas	%	39	(Brown et al., 2010)
Gas loss ratio	%	2	Valorgas[2]

[1] Centre de référence en agriculture et agroalimentaire du Québec (Ville de Saint-Hyacinthe, 2023)
[2] Valorgas, version W7f-waste, University of Southampton, Southampton, UK

Table 3.14. Dependent parameters for biogas purification modelling in anaerobic digestion

Dependent parameter	Unit	Formula	Value
Biogas purified	m^3	Biogas produced - biogas flared - biogas heat	8,055,911
Biomethane	m^3	Biogas purified × CH_4 ratio in biogas × (1-CH_4 losses)	538,441

Using these parameters, fugitive emissions of CO_2 and CH_4 emissions were calculated using Eq. 3 and 4 respectively:

$$\textit{Fugitive } CO_2 \textit{ emissions} = \textit{Biogas purified} \times CO_2 \textit{ ratio in biogas} \times \textit{Gas losses} \times CO_2 \textit{ density} \quad \textit{Eq. 3}$$

$$\textit{Fugitive } CH_4 \textit{ emissions} = \textit{Biogas purified} \times CH_4 \textit{ ratio in biogas} \times CH_4 \textit{ losses} \times CH_4 \textit{ density} \quad \textit{Eq. 4}$$

Table 3.15. Flows for biogas purification modelling in anaerobic digestion

Flow	Unit	Value	Ecoinvent flow name
Biomethane	m^3	4,717540	-
CO_2, non-fossil	kg	5,768350	*Carbon dioxide, non-fossil*
CH_4, non-fossil	kg	10,8897	*Methane, non-fossil*

Lastly, the production of biomethane is assumed to be utilized by the treatment facility to power the digestion process. As such, the heat requirement for this treatment facility is met by the biogas produced on-site. This not only avoids the need for external natural gas but also eliminates the associated background emissions related to its production life cycle. To model this offset process, the life cycle of producing and using natural gas was incorporated into the system but marked as "avoided" in OpenLCA. This amount of natural gas avoided by the process is given a negative value to reflect the emissions saved by utilizing biogas instead. The flow used was *"market for natural gas, high pressure | natural gas, high pressure | Cutoff, S."*

Table 3.16. Summary of the life cycle inventory of anaerobic digestion

Anaerobic digestion process			
	Input		
Flow	**Unit**	**Value**	**Source**
Sewage sludge	kg	13,043000	CVMO[1]
Sludge transport by freight	kg.km	10	Default value
Organic matter	kg	161,459000	CVMO
Organic matter transport by freight	kg.km	10	Default value
Electricity consumption	kWh	80,6188	Valorgas[2]
Heat consumption	MJ	348,0650	Valorgas
Digesters	item	0.02812	calculated
Digestate storehouse	item	0.02812	calculated
Gas holder	item	0.02812	calculated
	Output		
Biogas produced	m^3	9165970	CVMO
Digestate	kg	27429000	CVMO
CH_4, non-fossil	kg	72920,8	calculated
CO_2, non-fossil	kg	202,7270	calculated
Biogas purification			
	Input		
Chemical factory	item	0.00215	Ecoinvent 3.6
Electricity consumption	kWh	26,000000	Ecoinvent 3.6
	Output		
Biomethane	m^3	4,717540	calculated
CH_4, non-fossil	kg	10,8897	calculated
CO_2, non-fossil	kg	5,768350	calculated
Avoided natural gas production			
Natural gas	m^3	2455.3	calculated

[1] Centre de référence en agriculture et agroalimentaire du Québec (Ville de Saint-Hyacinthe, 2023)

[2] Valorgas, version W7f-waste, University of Southampton, Southampton, UK

3.4.2.3. Alkaline Stabilization

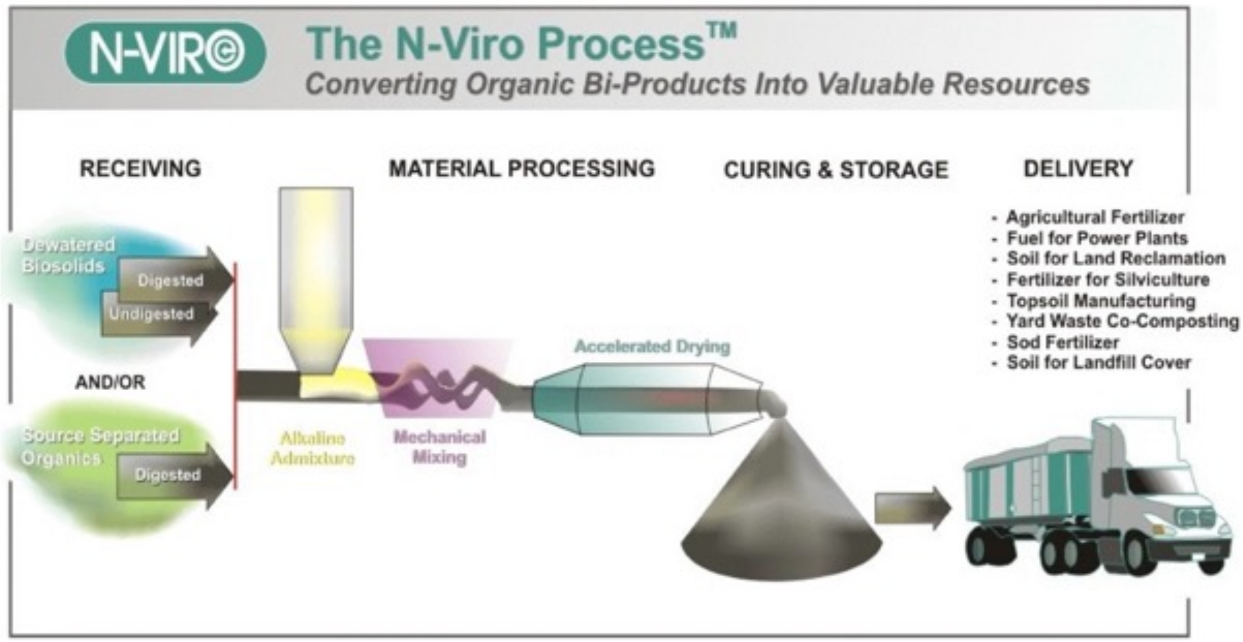

Figure 3.3. Alkaline stabilization (Halifax Regional Council, 2010) (used with permission from Walker Industries)

The process of alkaline stabilization of sewage sludge, as modeled in this study, is based on the N-Viro™ technology employed at the municipal treatment facility in Halifax, Nova Scotia. The inventory data was compiled from responses obtained through a questionnaire sent to the facility, the completed version of which is included in appendix 2 for reference.

In the facility's process, the biosolids received are conveyed to a mixing bin where an alkaline admixture is added. The facility uses cement kiln dust as the alkalizing agent, adding it at a rate of 35% on a wet weight basis of the biosolids input. To ensure sufficient free lime (CaO, $Ca(OH)_2$ or other strong alkali) in the admixture, quicklime (CaO) is also added at a rate of 1.5 kg per 100 kg of wet sludge input. The facility provided data on their diesel, electricity, and heat requirements. Additionally, they reported a water consumption rate of 110 m3/month, primarily used to hydrate the biofilter for ventilation.

Cement kiln dust, an industrial by-product of the cement production process, is purchased by N-Viro Systems from a cement factory. The cost of cement kiln dust was not disclosed. However, the fact that N-Viro Systems purchases cement kiln dust indicates that it is not considered as a residual from the cement industry, but rather a valuable, marketable product. Consequently, its production carries significant embedded supply chain (scope 3) carbon emissions (Brown et al., 2010). To account for the impact contributed by the use of cement kiln dust in ATB, an economic allocation was used. This method assigns environmental burdens between different products or processes based on their economic value, providing a more accurate reflection of the environmental impact associated with the use of cement kiln dust in the ATB process. The Ecoinvent 3.6 clinker production process ("*clinker production* | *clinker* | *Cutoff, S*") does not consider any coproduct apart the reference product 'clinker'. So, here the Ecoinvent process was modified in order to consider the cement kiln dust as a coproduct in addition to 1 kg clinker and to allocate it a part of the impact, based on an economic allocation rule. cement kiln dust is a by-product from clinker production. It is a very heterogeneous powder entrained in the combustion gasses flowing through the cement kiln and collected as residue in the air pollution control devices. Cement manufacturing process parameters, such as raw feeds, fuel characteristics and kiln technology largely influence the chemical composition and particle size of cement kiln dust. Although the majority of cement kiln dust is recycled back into the cement kiln as raw feed, limits on the amounts of alkalis and chlorides in cements call for further reuses of this waste. Hence, the remaining is sold for beneficial use or managed as a waste. The share of the net production of cement kiln dust that is actually sold for beneficial use was estimated to be 0.5 from 0-1. The cement kiln dust generation rate was 0.15 t/t clinker (Huntzinger & Eatmon, 2009). It was assumed that 67% of cement kiln dust generated is recycled back as feed in the kiln as the majority of the cement kiln dust is recycled (Adaska & Taubert, 2008) . Clinker was priced at 123.5 USD/t and cement kiln dust was priced at 30 USD/t in 2019 (U.S. Geology Survey, 2020). A summary of the parameters used for the economic allocation are displayed below in table 3.17:

Table 3.17. Parameters used for the economic allocation of cement kiln dust

Parameter	Unit	Value	Source
cement kiln dust beneficial use rate	0 to 1	0.5	Expert estimate
cement kiln dust generation rate	kg cement kiln dust/ kg clinker	0.15	(Huntzinger & Eatmon, 2009)

cement kiln dust recycling rate	kg cement kiln dust recycled/ kg cement kiln dust generated	0.67	(Adaska & Taubert, 2008)
cement kiln dust price	USD/ kg	0.03	(U.S. Geology Survey, 2020)
Clinker price	USD/ kg	0.1235	(U.S. Geology Survey, 2020)

Based on the parameters above, the net cement kiln dust production rate was first calculated. It is the net output of cement kiln dust for beneficial use and for waste management from the cement factory per kg of clinker produced. Next, the cement kiln dust beneficial use production rate which is the net output of cement kiln dust for beneficial use from the cement factory per kg of clinker produced was calculated. Then, using the same rate for cement kiln dust beneficial use production, the cement kiln dust revenue was calculate based on the cost displayed in the parameter table 3.17 above. The dependent parameters used for the economic allocation calculation are summarized in the table 3.18 displayed below:

Table 3.18. Dependent parameters used for the cement kiln dust economic allocation in alkaline stabilization

Dependent parameters	Unit	Formula	Value
cement kiln dust net production rate	kg cement kiln dust/kg clinker	cement kiln dust generation rate x (1-cement kiln dust recycling rate)	0.049
cement kiln dust beneficial use production rate	kg cement kiln dust/kg clinker	cement kiln dust net production rate × cement kiln dust beneficial use share	0.024
cement kiln dust revenue	USD	cement kiln dust beneficial use production rate × cement kiln dust price	0.00074
Clinker revenue	USD	Clinker price × 1	0.1235
Clinker economic allocation		Clinker revenue/ (Clinker revenue +cement kiln dust revenue)	0.994

cement kiln dust economic allocation	cement kiln dust revenue/ (Clinker revenue +cement kiln dust revenue	0.00598

Finally, the listed cement kiln dust and clinker economic allocation values are input into OpenLCA in the Allocation window for economic allocation. As for the flows of cement kiln dust and clinker they were given their economic value based on their respective revenues (listed in table 3.18 above) and presented again in the table below with their respective Ecoinvent flow names:

Table 3.19. Flows for the economic allocation of cement kiln dust in alkaline stabilization

Flow	Unit	Value	Ecoinvent flow name
cement kiln dust	USD	0.00074	*cement kiln dust*
Clinker	USD	0.12350	*clinker*

Table 3.20. Summary of the life cycle inventory of alkaline stabilization

ATB process			
	Input		
Flow	**Unit**	**Value**	**Source**
Sewage sludge	t	10,850	N-Viro*
Sludge transport by freight	kg.km	10	Default value
Diesel	Mj	102,000	N-Viro
Quicklime, milled loose	t	465	N-Viro
cement kiln dust	t	10,850	N-Viro
Electricity consumption	kWh	72,000	N-Viro
Heat consumption	Gt	26,400	N-Viro
Tap water	kg	1,316,040	N-Viro
	Output		
Alkaline stabilized biosolids	t	34,000	N-Viro
Water	kg	1,316,040	N-Viro

* See appendix 7.2

3.4.2.4. Composting

The modelling of the composting process was based on the operations of Gaudreau Environnement Inc., located in Victoriaville, Quebec. The inventory data required for this process was gathered from a variety of sources, including the company's website, personal communications with the company, and existing literature on LCA studies related to composting. Additionally, the Biosolids Emissions Assessment Model (BEAM), developed by the Canadian Council of Ministers of the Environment (CCME), was utilized for composting-specific data.

The composting process at the Gaudreau facility employs a windrow composting technique, which involves a mixture of 40% sewage sludge and 60% yard waste, such as wood and sawdust. To account for the infrastructure used in windrow composting processes, the "composting facility, open" flow from the Ecoinvent 3.6 database was used. The quantity was calculated based on the lifetime capacity of the composting facility, which was assumed to be 30 years in this case. Sawdust was also included in the model as "sawdust, loose, wet, measured as dry mass" and is used at a rate of 39 t/day (Sylvis, 2009).Diesel consumption for machine operation, represented as "machine operation, diesel, >= 74.57 kW, low load factor," was provided by the BEAM model at a rate of 697 L/day (Sylvis, 2009). The machine operation consumes 5.95 kg of diesel per hour of operation, as per the Ecoinvent 3.6 database. The methodology for output flows incorporated emissions of CO_2, CH_4, and N_2O from compost piles. The BEAM model provided a default input value for sewage sludge at 100,000 kg and indicated that approximately 40-70% of this quantity is converted into compost. For the purposes of this study, an average conversion rate of 55% was adopted. The parameters under consideration are detailed in Table 3.21 below:

Table 3.21. Parameters for compost production

Parameter	Unit	Value	Source
Sewage sludge input	kg	100,000	BEAM* default
Compost output	%	55 (40-70)	BEAM model
Diesel consumption	L/day	697	BEAM model
Diesel density	kg/L	0.85	(Speight, 2011)
Diesel time operation	kg/h	5.95	Ecoinvent 3.6 database
Saw dust	t/day	39	BEAM calculation

CH_4 compost pile emissions	kg-CH_4	339	BEAM calculation
N_2O compost pile emissions	kg-N_2O	24	BEAM Calculation
CO_2 ratio in compost	kg/kg compost	0.22	BEAM model

[*] Biosolids Emissions Assessment Model

The compost produced from this process is hence calculated using the following parameters:

Table 3.22. Dependent parameter for compost production calculation

Dependent parameters	Unit	Formula	Value
Compost produced	kg	Compost output × sewage sludge input	55,000

Table 3.23. Summary of the life cycle inventory of composted biosolids

COM process[1]			
		Input	
Flow	**Unit**	**Value**	**Source**
Sewage sludge	kg	55,000	BEAM[2] default
Sludge transport by freight	kg.km	10	Default value
Machine operation by diesel	h	99.4	Ecoinvent 3.6
Composting facility	item	0.00041	Own calculations
Saw dust	kg	39	BEAM calculations
		Output	
Composted biosolids	kg	55,000	BEAM calculations
CO_2, non-fossil	kg	12,100	Own calculations
N_2O	kg	24	BEAM calculations
CH_4	kg	330	BEAM calculations

[1] All quantities are calculated on a per day basis.
[2] Biosolids Emissions Assessment Model

3.4.3. Life Cycle Inventory for Urea Production

In this study, urea fertilizer was utilized for silage corn cultivation. The application was either at a 100% rate to meet the Functional Unit (functional unit) of 120 kg-N/ha required for corn silage cropping, or in combination with biosolids at a 1:1 ratio. As detailed in section 3.1.1 (Description of the Montreal experimental design), the treatments receiving 100% N from urea incorporated 39 kg-N/ha as starter fertilizer, and 81 kg-N/ha as urea. Given that urea contains 46% N (46-0-0) (CRAAQ, 2013), the amount of urea application needed to meet the functional unit of the study equates to 176 kg of urea for full application rate and 88 kg for a 1:1 combination ratio with biosolids. The process flow utilized in OpenLCA for this operation was *"market for urea, as N | urea, as N | Cutoff, U "* from the Ecoinvent 3.6 database. The transportation of urea from the production site to the farm was also factored into the model, with an assumed truck travel distance of 20 km to the farms.

Table 3.24. Parameters for urea production and transportation to farm

Parameter	Unit	Value	Source
Urea 100% application rate	kg-N	81	Experimental design
Urea 50% application rate	kg-N	40.5	Experimental design
Transportation distance to farm	km	20	Default value

3.4.4. Life Cycle Inventory for Agricultural Application of Biosolids

3.4.4.1. Transportation of Biosolids to the Field

Treated sewage sludge, now called biosolids, are assumed to be applied on the same agricultural field to simplify the comparison. Transportation distance from treatment facility to the farm is assumed to be 20 km. Transportation is carried out by diesel trucks with a carrying capacity 7.5-16 tons (Poulsen & Hansen 2003). The transportation process is modelled in OpenLCA as *"market for transport, freight, lorry 7.5-16 metric ton, EURO5 | transport, freight, lorry 7.5-16 metric ton, EURO5 | Cutoff, S"* (Ecoinvent, 2019).

3.4.4.2. Land Application on the Field

The modeling of land-application for both biosolids and urea fertilizers incorporates inputs related to the use of agricultural machinery, land use, and the quantity of fertilizer required. On the output side, the model accounts for the GHG emissions that emanate from the field following the application of these materials. As outlined in section 3.1.2, both biosolids and urea fertilizers are either surface-spread (SS) or incorporated (INC) by cultivation, to examine the effect of incorporation on GHG emissions. When biosolids were surface-spread, a hydraulic loader and spreader for solid manure *("solid manure loading and spreading, by hydraulic loader and spreader | solid manure loading and spreading, by hydraulic loader and spreader | Cutoff, S "*) was employed to distribute the biosolids on the land. In cases where biosolids were incorporated, the same hydraulic loader was used, in addition to a rotary cultivator for tillage *("tillage, rotary cultivator | tillage, rotary cultivator | Cutoff, S ")* across an area of 1ha, to blend the top layer of the soil with the fertilizer. Urea was applied using a fertilizer broadcaster *("fertilising, by broadcaster | fertilising, by broadcaster | Cutoff, S ")* over an area of 1ha. Other flows common to all scenarios include land occupation of 1 ha/year *("Occupation, annual crop, CA")*, the use of 39 kg-N of starter fertilizer ammonium nitrate *("ammonium nitrate production | ammonium nitrate, as N | Cutoff, S ")*, and a harvesting flow that combines equipment and machine operations on the field over an area of 1 ha.

3.4.4.3. GHG Emissions from the Field

Output emissions of CH_4, CO_2, and N_2O, were measured from the field using gas sampling from each plot during the growing seasons from 2017 to 2019. Each experimental treatment had 4 replicates.

The sampling was conducted using manual non-steady-state vented chambers at the Macdonald Research Farm, affiliated with McGill University (Obi-Njoku et al., 2023; Obi-Njoku et al., 2022). The field measurements were collected over the three-year period of the study (2017-2019), but only the data collected from 2018 and 2019 were used in this analysis because the data from 2017 were incomplete. While all three principal GHGs were present in the field measurements, only N_2O emissions were included in the carbon footprint assessment because CH_4 emissions were not significant, and (biogenic) CO_2 emissions are not reported in GHG inventories. The first step was to perform a statistical analysis to compare the data measured for 2018 and 2019. The variables were processed using the paired sample T-test to determine whether the mean difference between the two sets of observations for 2018 and 2019 was significant at $\alpha = 0.05$. Two-tailed paired t-tests were performed for 2018 and 2019 for CO_2 emissions, N_2O emissions, total N concentration and silage yield. The paired T-test was performed using Microsoft Excel

(2013, Microsoft, Redmond, WA). The paired T-test showed that the datasets for both years were statistically different: CO_2 emissions (P is less than 0.001), N_2O emissions (P = 0.018), silage yield (P is less than 0.001). As a result of this statistical analysis, the data will be processed and analyzed as two separate sets of data. This study integrates the experimental data collected from the year 2018.

The CO_2 and N_2O emissions dataset from 2018 were exported to Microsoft Excel and displayed in function of date of sampling and treatment plot. The emissions are in g/ha/day. The first step was to calculate the mean by treatment of the four replicates. This was done using "AVERAGEIF" function in Excel. After the mean by treatment was generated, the cumulative flux in g/ha was calculated using Eq. 5 below:

$$F = (t1 - t0) \times (C1 + C0)/2 \qquad \text{Eq. 5}$$

F is the cumulative gas flux of CO_2 or N_2O (g/ha), t is the time in days (d) of sampling and C is the sample mass concentrations (g/ha/day) of CO_2 or N_2O at time t (d).

The cumulative emissions of CO_2 and N_2O in kg/ha by treatment are presented in the table 3.25 below:

Table 3.25. Cumulative N_2O and CO_2 emissions of land-applied biosolids by treatment from 2018

Treatments	N_2O cumulative emissions (kg/ha)	CO_2 (biogenic) cumulative emissions (kg/ha)
Control zero	0.56	7262.8
UF_SS	5.28	8062.8
UF_ INC	3.5	6908.1
AD_SS	7.6	9629.6
AD_INC	10.9	10025.9
ATB_SS	1.9	10824.9
ATB_INC	1.4	9240.7
COM_SS	0.9	12538.5
COM_INC	0.6	12830.1
UF+AD_SS	7.7	8669.7
UF+AD _INC	9.1	10344.7
UF+ATB_SS	1.9	9931.4
UF+ATB_INC	1.5	8652.3
UF+COM_SS	3.4	10764.8
UF+COM_INC	2.8	9450.8

UF: urea fertilizer; AD: anaerobic digestion; ATB: alkaline treated biosolids; COM: compost; SS: surface-spread; INC: incorporated

3.4.5. Avoided Sewage Sludge Incineration Scenario

In order to account for the emissions avoided by not incinerating sewage sludge, but instead treating and applying it to land, the system boundary was expanded to include the avoided process of incinerating an equivalent quantity of sewage sludge. This was achieved by incorporating the *"biowaste"* flow from the process *"treatment of biowaste, municipal incineration | biowaste | Cutoff, S"* in OpenLCA, which represents the activity of disposing of biowaste in a municipal solid waste incinerator. The quantity of sewage sludge that avoided incineration was calculated using Eq. 6:

$$\textit{Avoided sludge incinerated} = (\textit{Sewage sludge input} \times \textit{biosolids applied on land})/\ \textit{biosolids produced} \qquad \textit{Eq. 6}$$

The amount of sewage sludge for each scenario is presented in the table 3.26 below:

Table 3.26. Avoided sewage sludge disposal by incineration across different life cycle scenarios

Treatments	AD	ATB	COM	UF+AD	UF+ATB	UF+COM
Avoided sewage sludge disposal by incineration (kg)	6,788	23,823	162,329	3,394	11,911	81,164

AD: anaerobic digestion; ATB: alkaline treated biosolids; CO: compost; UF: urea fertilizer

3.4.6. Avoided Sewage Sludge Landfilling Scenario

As the Ecoinvent database does not include a process for the landfilling of raw sewage sludge, a life cycle inventory calculation tool developed by Gobar Doka was used to generate this process (Doka, 2021). Doka developed a life cycle inventory model of regionalised waste treatment. The tool was created to calculate waste-specific inventories of waste disposal. Process inventories of advanced disposal technologies like municipal incineration and sanitary landfilling are possible, but also more rudimentary disposal technologies like open burning and open dumping are possible. The underlying process models are built upon the existing models already used for waste disposal in Ecoinvent and the updates. The tools allow calculation of inventories and produce Ecospold2 XML-Files (ES2) which can be read into EcoEditor and via that route become part of the Ecoinvent database. The calculation tools are made up by several Excel workbooks however for the landfilling modelling, Central Repository 2020.xlsm workbook was chosen. Next, the landfilling dataset was selected, and its number was input in cell B12. Then a treated waste material was chosen from the list in this case "sewage sludge 2016". For disposal side/ geographic location, Copper Canada 2015 was used, and the temperatures were changed to Montreal climate (local conditions). "Sanitary landfill" was used as treatment/ disposal type. After inputting all the correct entry information, the file was created by changing the number in cell B12 by the number of the cell for the targeted activity and then selecting the grey cell "save ES1". This created the Ecospold2 XML file that was then imported to the Ecoinvent database on OpenLCA. The newly created flow, titled "disposal, raw WWTP sludge, to sanitary landfill/CA U," represents the quantity of raw sewage sludge being diverted away from the landfilling process. These emissions are negative emissions since it is an avoided process. The sewage sludge quantity being diverted from the landfill was calculated for each scenario using Eq. 7:

$$\textit{Avoided sludge landfilled} = (\textit{Sewage sludge input} \times \textit{biosolids applied on land})/\textit{ biosolids produced} \qquad \textit{Eq. 7}$$

The amount of sewage sludge for each scenario is presented in the table 3.2.6.1 below:

Table 3.27. Avoided sewage sludge disposal to landfill across different life cycle scenarios

Treatments	AD	ATB	COM	UF+AD	UF+ATB	UF+COM
Avoided sewage sludge disposal to landfill (kg)	6,788	23,823	162,329	3,394	11,911	81,164

AD: anaerobic digestion; ATB: alkaline treated biosolids; CO: compost; UF: urea fertilizer

3.5. Life Cycle Impact Assessment

For the life cycle impact assessment, the IPCC methodology was employed. This study concentrated exclusively on the climate change impacts over a 100-year horizon (GWP100), quantified in kg-CO_2e. The subsequent table 3.28 delineates the GWP100 of the 15 scenarios assessed within this research.

Table 3.28. Processes contributing to GWP100 (kg-CO_2e)

Scenarios	Treatments	Sewage sludge transport	Fertilizer treatment	Feedstock transport	Fertilizer transport	Land application	Avoided products	Net total (kg-CO_2e)
1	Ctr zero					1270.4		1,270.4
2	UF_SS		270.7		0.79	2058.4		2,329.9
3	UF_ INC		270.7		0.79	1591.9		1,863.4
4	AD_SS	15.2	3390	189.1	64.2	2832.3	-2164.3	4,326.7
5	AD_INC	15.2	3390	189.1	64.2	3867.4	-2164.3	5,361.9
6	ATB_SS	53.6	4013.7		117.6	1212.7		5,397.7
7	ATB_INC	53.6	4013.7		117.6	1122.3		5,307.3
8	COM_SS	365.3	33932.6		401.9	1362.9		36,062.9
9	COM_INC	365.3	33932.6		401.9	1323.7		36,023.7
10	UF+AD_SS	7.6	1695	94.5	32.13	3018.8	-1082.1	3,766.1
11	UF+AD_INC	7.6	1695	94.57	32.13	3478.8	-1082.1	4,226
12	UF+ATB_SS	26.8	2006.9		58.8	1294.4		3,387
13	UF+ATB_INC	26.8	2006.9		58.8	1233.7		3,326.3
14	UF+COM_SS	182.6	16966.1		200.9	1939.6		19,289.4
15	UF+COM_INC	182.6	16966.1		200.9	1804.1		19,153.9

UF: urea fertilizer; AD: anaerobic digestion; ATB: alkaline treated biosolids; COM: compost; SS: surface-spread; INC: incorporated

Chapter 4 Results and Discussion

The results discussed below represent the contribution of each scenario to short-term climate change when providing the functional unit of the study, which is 120 kg-N/ha for corn silage crops.

Scenario 1: Negative control

Figure 4.1 depicts the contribution per life cycle stage of the negative control scenario (no fertilizer) compared with the scenarios in which urea is surface-spread or incorporated. The emissions from a plot without fertilizer application are indicative of background emissions from inherent soil processes. This control plot has a quantified climate change score (GWP100) of 1270.4 kg-CO_2e, serving as a reference for subsequent treatments. Emissions exceeding this baseline are attributed to the land application of biosolids.

Scenarios 2 and 3: Urea (surface-spread and incorporated)

In the same figure 4.1, the life cycle analysis of urea production and application to agricultural soil has a climate change contribution of 1863.4 kg-CO_2e for incorporated urea and 2329.9 kg-CO_2e for surface-spread urea. Notably, 85% (for incorporated) and 88% (for surface-spread) of this impact is attributed to the land application phase. The predominant factor in this

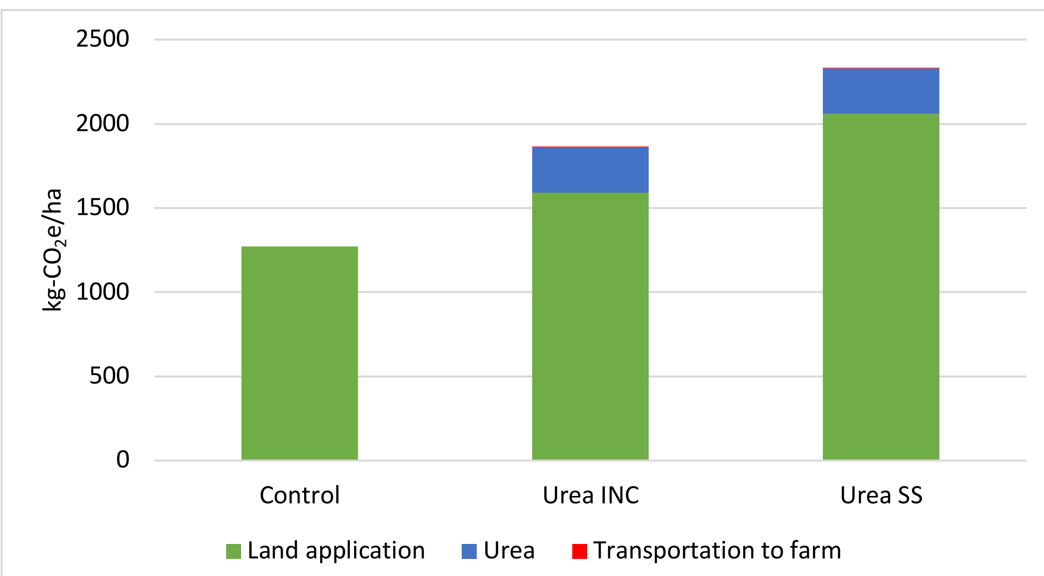

Figure 4.1. Life cycle contribution of urea production and land application to climate change (GWP100)

phase is N_2O emissions, accounting for 76.5% (1575 kg-CO_2e for surface-spread) and 66.3% (1056 kg-CO_2e for incorporated) of the total climate change score from land application.

Scenarios 4 and 5: Digestate (surface-spread and incorporated) and;

Scenarios 10 and 11: Half digestate and half urea (surface-spread and incorporated)

Figure 4.2 presents the life cycle contribution to climate change of digestate, and urea production followed by land application. The anaerobic digestion scenario followed by land application accounted for a net score of 4326.7 kg-CO_2e for surface-spread fertilizers and 5361.9 kg-CO_2e for incorporated fertilizers. For the plots that received half of the N requirement from biosolids and half from urea, the contribution was 4226 kg-CO_2e for incorporated and 3766.1 for surface-spread. Emissions from urea production for both 50:50 scenarios included in the land application stage and account for 133 kg-CO_2e. The land application stage of digested biosolids comprises the biggest part of the climate change contribution in all four scenarios. Direct emissions

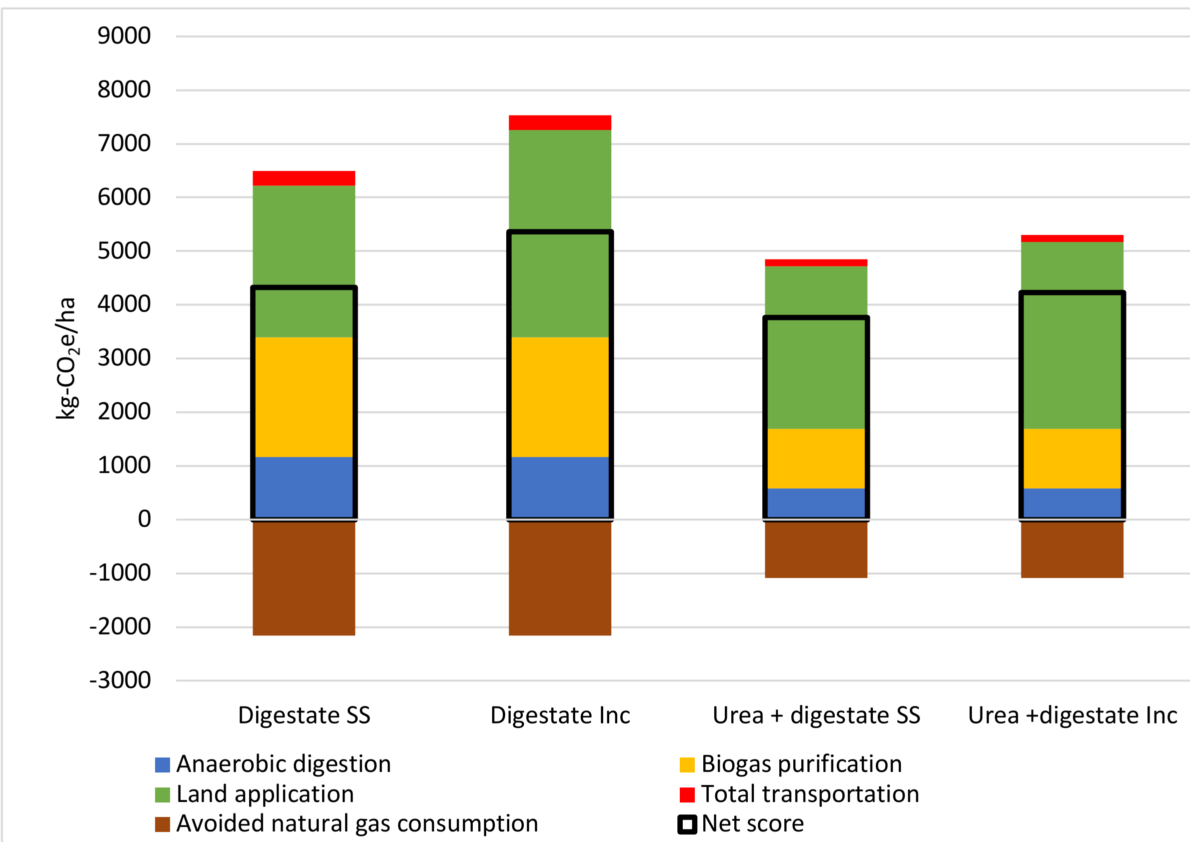

Figure 4.2. Life cycle contribution of digestate production and land-application to climate change (GWP100)

from land application include CH_4 and N_2O emissions. CO_2 emissions coming from the land application of biosolids are considered biogenic thus have no impact on climate change. However, CO_2 emissions from the land application of urea are considered fossil and thus have an impact. Negative emissions are the avoided consumption of natural gas due to the co-production of biogas during anaerobic digestion of sewage sludge. As can be noted, around 2000 kg-CO_2e were avoided due to the biogas production in anaerobic digestion. For the blend of urea and digested biosolids, half of the previous amount was avoided due to the treatment and production of only half the digestate needed to meet the functional unit.

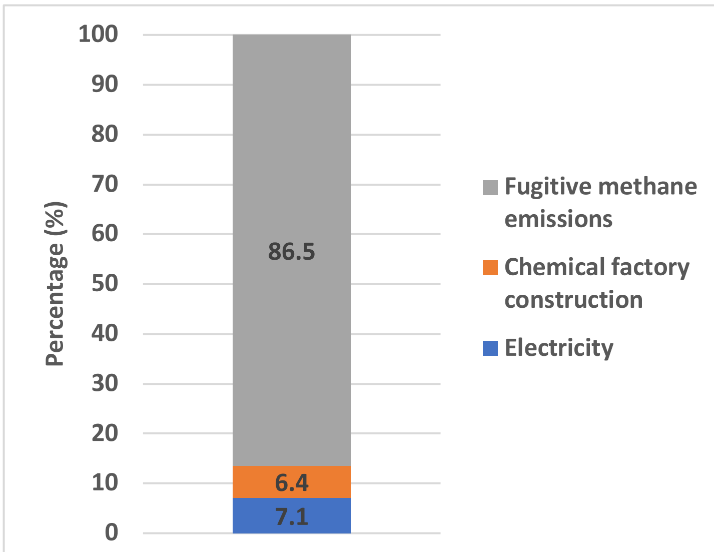

Figure 4.3. Percentage contribution of anaerobic digestion to climate change (GWP100)

Subsequent to land application, which is responsible for nearly 80% of the total climate change score, the anaerobic digestion process contributes an additional 580.6 kg-CO_2e, representing 15.4% of the overall climate change contribution. As can be seen in figure 4.3, most of this impact (96.6%) is attributed to fugitive methane emissions during digestion. This was computed based on the premise that 2% of the biogas generated during the digestion process is lost (Brown et al., 2010).

Furthermore, biogas purification accounts for 29.6% of the overall climate change score, resulting in emissions of 1114.3 kg-CO_2e. Similar to the digestion process, figure 4.4 shows that the predominant contributor to the score during biogas purification is fugitive methane emissions (86.5%), estimated to be 4% at this stage (Brown et al., 2010).

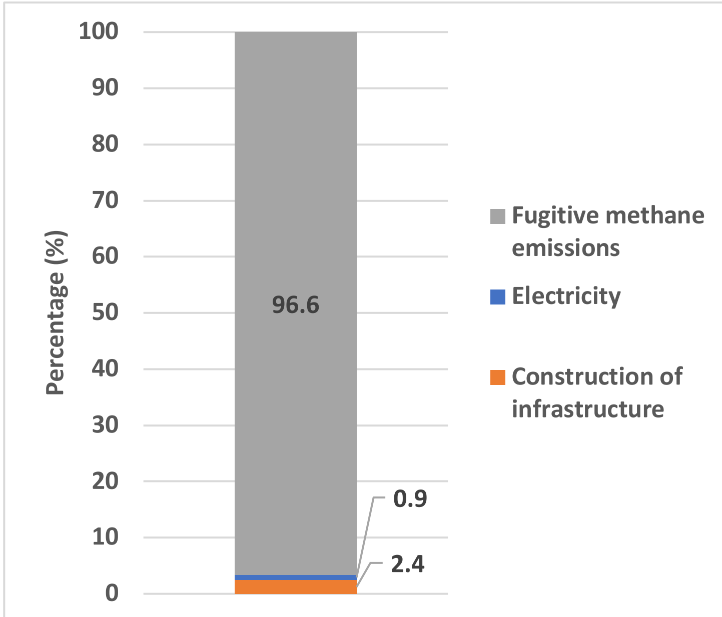

Figure 4.4. Percentage contribution of biogas purification to climate change (GWP100)

Scenarios 6 and 7: Alkaline biosolids (surface-spread and incorporated) and;

Scenarios 12 and 13: Half alkaline biosolids and half urea (surface-spread and incorporated)

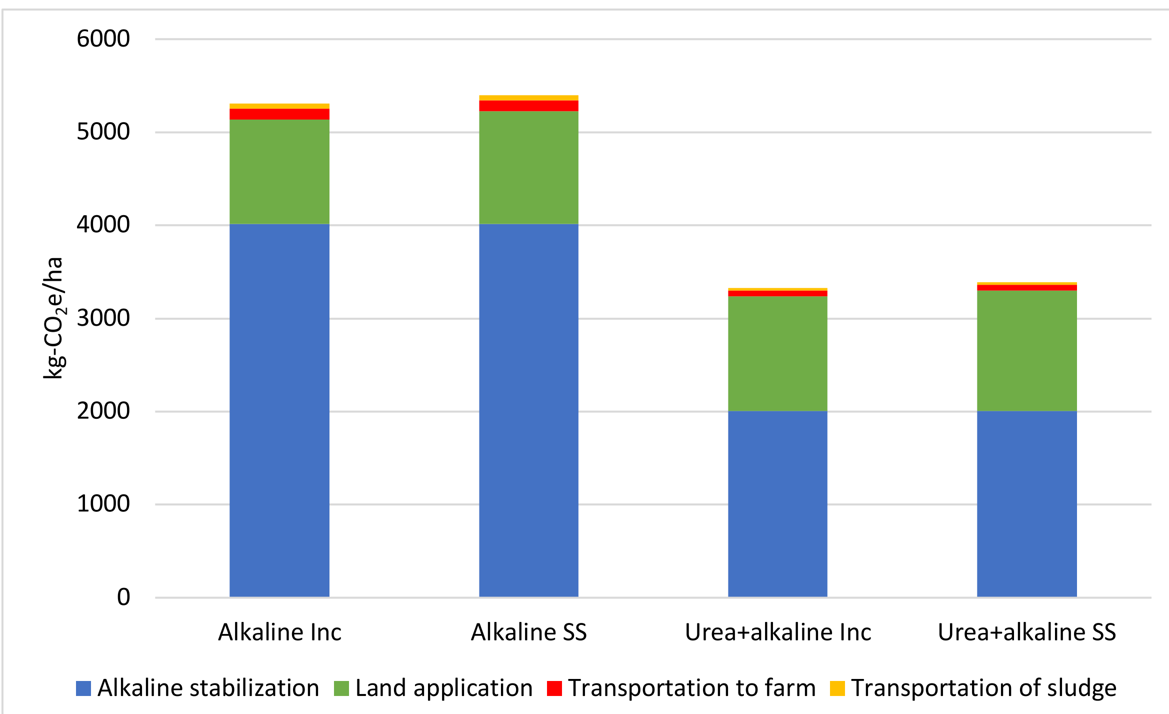

Figure 4.5. Life cycle contribution of alkaline stabilization and land application to climate change (GWP)

Figure 4.5 represents the contribution to climate change of alkaline stabilization and urea production scenarios, per life cycle stage. The alkaline stabilization of biosolids, followed by land application, resulted in an impact of 5397.7 kg-CO_2e for surface-spread and 5307.3 kg-CO_2e for incorporation. For plots that received half of their N requirement from biosolids and the other half from urea, the impact was 3326.3 kg-CO_2e for incorporated and 3387 kg-CO_2e for surface-spread. Unlike the scenarios previously examined, in these four scenarios, the alkaline stabilization treatment process represents the largest contribution to climate change and accounts for 75% of this contribution.

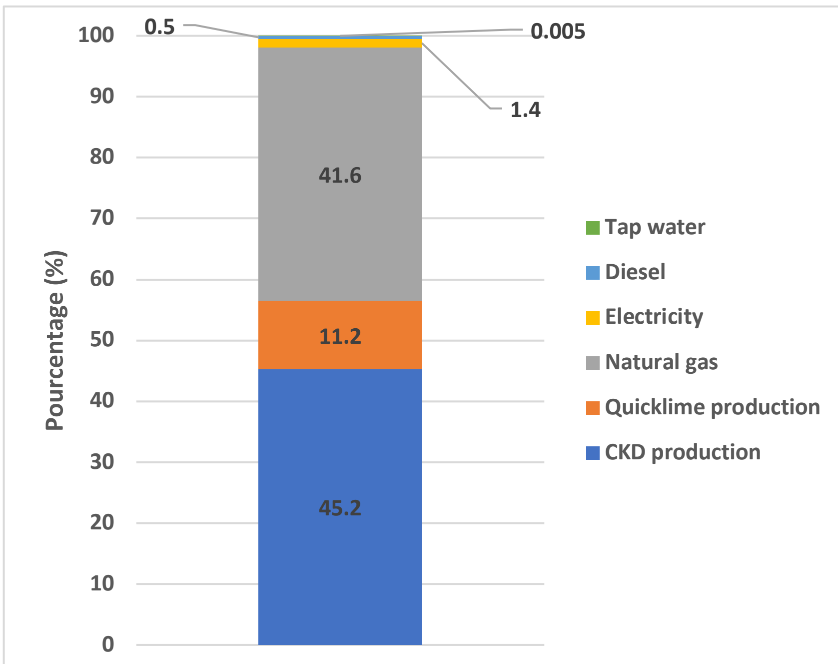

Figure 4.6. Percentage contributions to climate change of factors in the alkaline stabilization process (GWP100) (CKD: Cement kiln dust).

Upon analysis of the alkaline stabilization process, it is observed in figure 4.6 that 45.2% of the contribution stems from the production of cement kiln dust, which was calculated using an economic allocation, while 41.6% is attributed to natural gas consumed during treatment. Furthermore, the production of quicklime utilized in the process accounts for an additional 11% of the impact.

Scenarios 8 and 9: Compost (surface-spread and incorporated) and;

Scenarios 14 and 15: Half compost and half urea (surface-spread and incorporated)

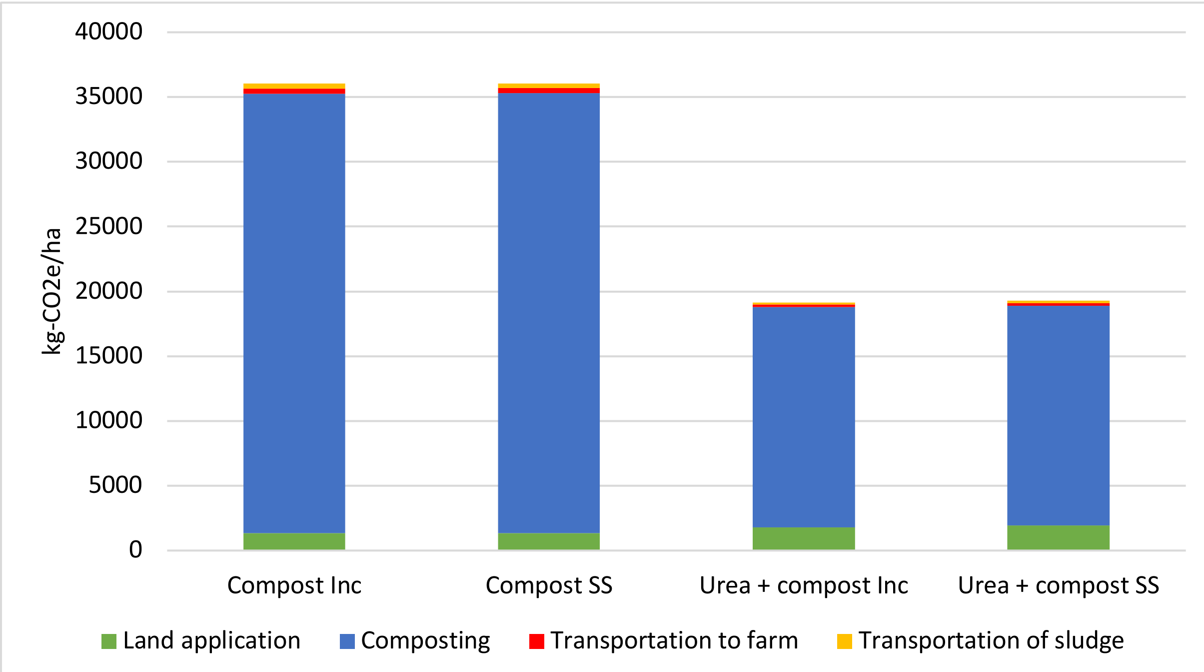

Figure 4.7. Life cycle contribution of compost production and land application to climate change (GWP100)

In figure 4.7, the life cycle contribution of composting biosolids followed by land application is presented in this graph. For the composting of biosolids followed by land application, the impacts were quantified at 36,062 kg-CO_2e for surface spreading and 36,023.7 kg-CO_2e for incorporation. In scenarios where plots received a 50:50 mix of N from biosolids and urea, the impacts were 19,153.9 kg-CO_2e and 19,289.4 kg-CO_2e for incorporated and surface-spread, respectively. Notably, the composting process itself accounted for 94% of the total impact. Within this composting contribution, CH_4 emissions from the compost pile were the predominant factor, constituting nearly half of the climate change impact. N_2O emissions from compost piles followed, contributing 34.2%. The remaining impact was attributed to emissions from diesel-fueled machinery operations.

The land application of composted biosolids accounted for 3% of the total contribution to climate change for these scenarios. Despite being only 3% of the total impact, land application still resulted in emissions of 1,323.7 kg-CO_2e for incorporated compost and 1,362.9 kg-CO_2e for surface-spread compost. Figure 4.9 showcases the percentage contribution of land application of compost to climate change. Within this segment, the process of solid manure loading and

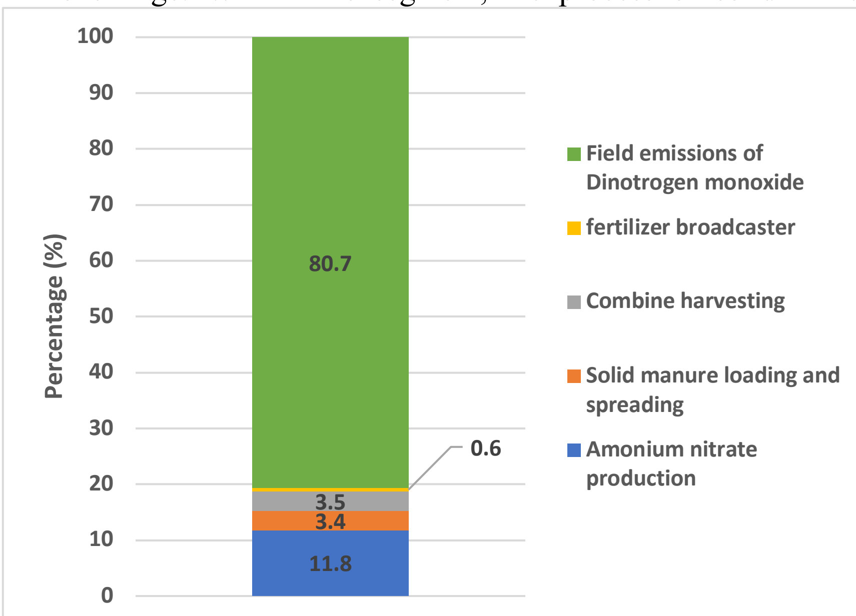

Figure 4.8. Percentage contribution to climate change of factors in the composting process (GWP100)

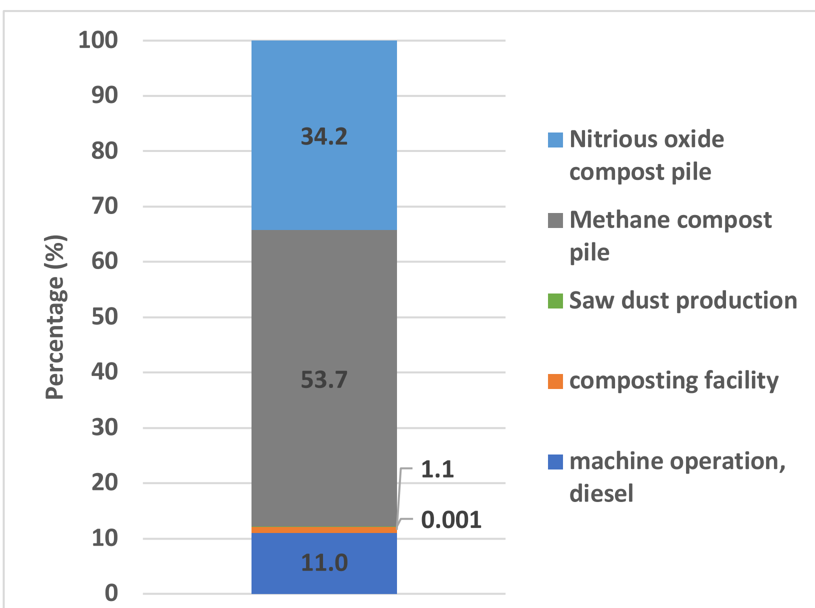

Figure 4.9. Percentage contribution to climate change of factors in the land application process (GWP100)

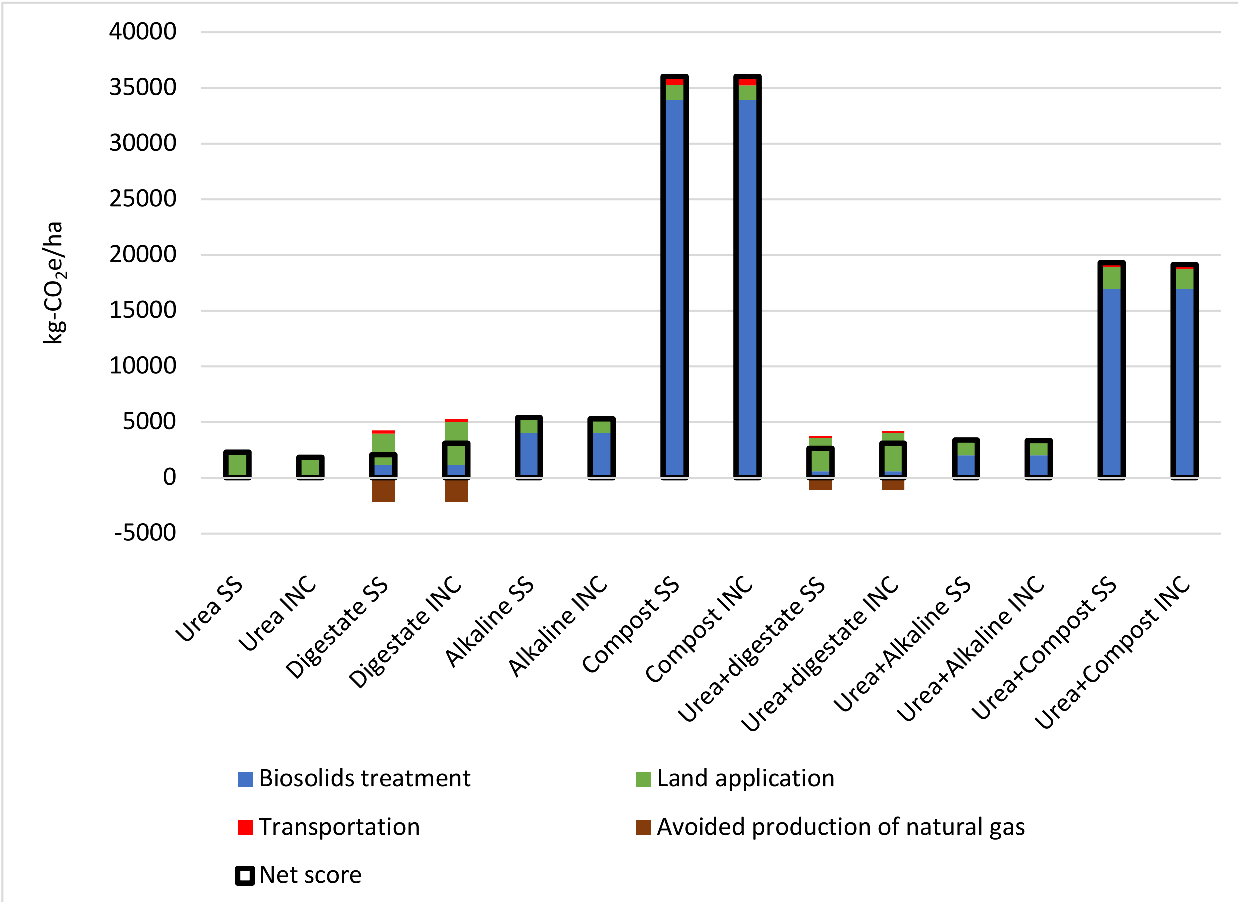

Figure 4.10. Comparative analysis of the life cycle impacts of the 15 different scenarios evaluated, all aiming to provide a functional unit of 120 kg-N/ha with respect to short-term climate change (GWP100)

spreading, employed to apply the biosolids to the land, was responsible for 45% of the emissions, equivalent to 618 kg-CO_2e. This emission level is 3.4 times greater than that for the land application of alkaline biosolids and nearly six times that of digestate application. The elevated emissions associated with solid manure loading and spreading can be attributed to the substantial volume of compost required for application. Given the lower nitrogen content in compost (0.004 kg-N/kg biosolids) compared to alkaline (0.01 kg-N/kg) and digestate (0.055 kg-N/kg) biosolids, a considerably larger volume of fertilizer was managed by the agricultural machinery, resulting in increased fuel consumption. Similarly, compost transportation to the farm resulted in notably higher emissions of 401.9 kg-CO_2e, compared to the 117 kg-CO_2e for alkaline biosolids and 64.26 kg-CO_2e for digestate. The disparity in emissions here was attributed again primarily to the larger volume of compost required to meet the requirements of the functional unit.

The 15 different scenarios were compared to assess their impact on climate change (figure 4.10). It is important to restate that all these scenarios were compared on the same basis, which was to provide 120 kg-N/ha to grow corn. Within the system boundary described, urea production followed by land application had the lowest impact on climate change, emitting 1863.4 kg-CO_2e for incorporated urea and 2329.9 kg-CO_2e for surface-spread urea. The half and half scenarios of digestate and urea production and land application (UF+AD SS and INC) followed with the second lowest contribution to climate change with a net total of 3766.1 kg-CO_2e for surface-spread and 4226 kg-CO_2e for incorporated. The negative emissions for all the treatments involving anaerobic digestion were due to the avoided production and consumption of natural gas due to the production of biogas during the digestion process. Research by Brown et al. (2010); Hospido et al. (2010); Houillon and Jolliet (2005); Murray et al. (2008); Poulsen and Hansen (2003) all found that anaerobic digestion scenarios perform well in regard to climate change mitigation compared to other treatment techniques, due to biogas co-production. On the other hand, scenarios involving sewage sludge composting exhibited the most pronounced contributions to short-term climate change. Emissions peaked at 36,062.9 kg-CO_2e for surface-spread compost and 36,023.7 kg-CO_2e for incorporated compost. The elevated GHG emissions from these composting scenarios predominantly stemmed from the processing phase. Specifically, composted biosolids followed by land application scenarios emitted GHGs at a rate 8 times higher than that of digested biosolids production and land application, and 6 times higher than alkaline biosolids production and land application.

It is crucial, however, to underscore that by converting sewage sludge into fertilizers for agricultural use, the sludge is diverted from other disposal methods, notably incineration and landfilling. This study incorporated both the avoided incineration and landfilling of sewage sludge into the system boundary to evaluate the environmental implications of opting for biosolids production over traditional disposal methods. Figure 4.11 represents the comparative analysis of the life cycle impacts of the 15 different scenarios with the account of the avoided incineration of sewage sludge. When accounting for the emissions avoided through incineration, the digestate scenario avoided 262.76 kg-CO_2e, the alkaline stabilization scenario avoided 922.12 kg-CO_2e, and the compost scenario avoided 6283.1 kg-CO_2e. The higher the volume of sewage sludge used during the process of biosolids treatment, the higher the emissions avoided from incineration. After subtracting these avoided emissions from the life cycle contribution of each scenario, the digestate

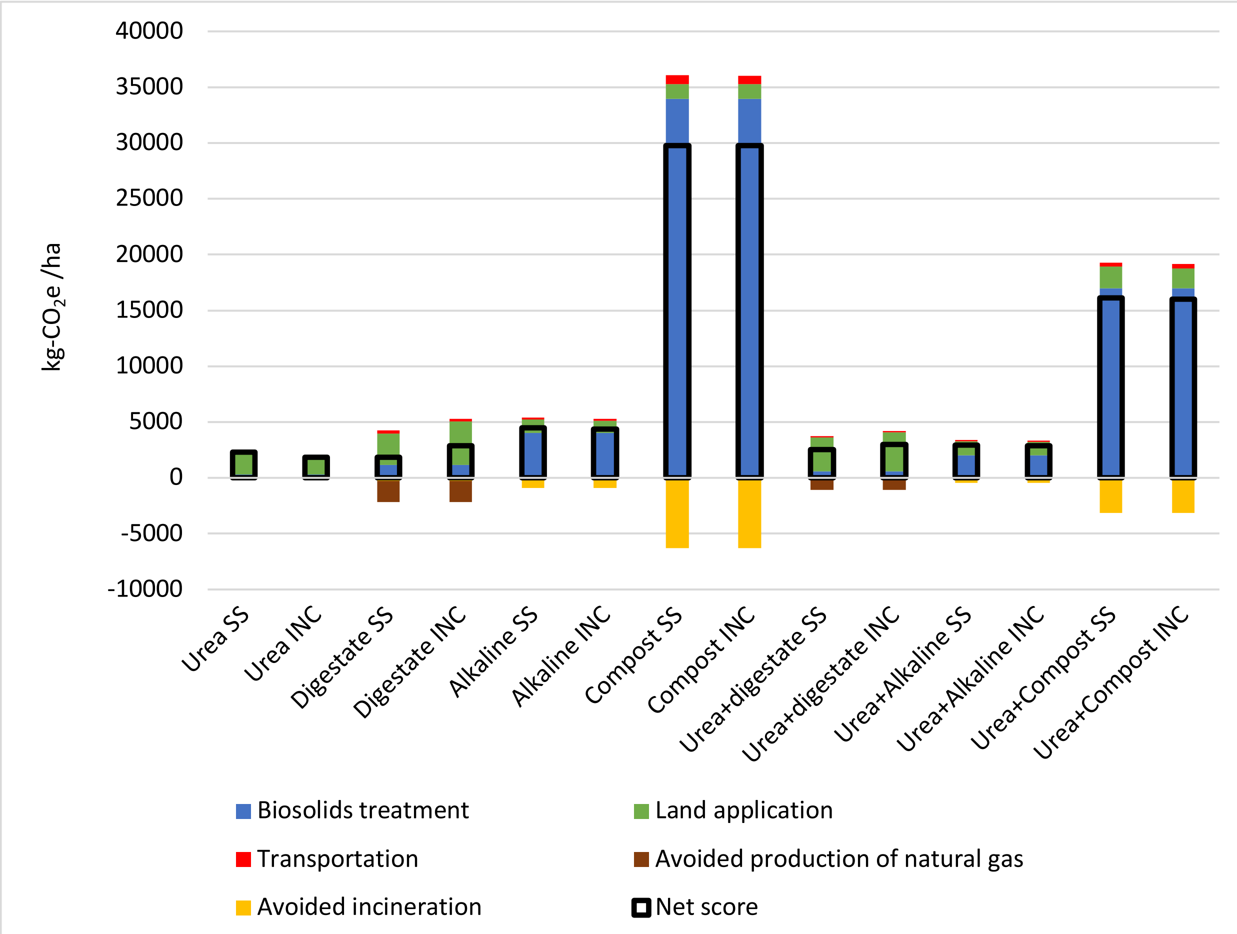

Figure 4.11. Comparative analysis of the life cycle impact of the 15 different scenarios, accounting for the avoided incineration of sewage sludge

production and land application emerged as the scenario with the least contribution to climate change, registering at 1835 kg-CO_2e. This is closely followed by the urea production and land application scenario at 1863.40 kg-CO_2e. Despite having the highest reductions, composted biosolids scenarios still remain the highest contributors to climate change, with contributions decreasing to 29,779.7 kg-CO_2e for surface-spread and 29,740.55 kg-CO_2e for incorporation. Among all scenarios assessed in relation to short-term climate change, anaerobic digestion followed by land application has the least environmental impact.

Landfilling of sewage sludge was also evaluated as an alternative disposal method for comparison to the land application scenarios. This comparative analysis is displayed in figure 4.12. For the digestate scenario, 5289 kg-CO_2e were avoided, while the alkaline stabilization and composted scenarios avoided 18,560 kg-CO_2e and 126,500 kg-CO_2e, respectively. After accounting for these avoided emissions, the compost production and land application scenario

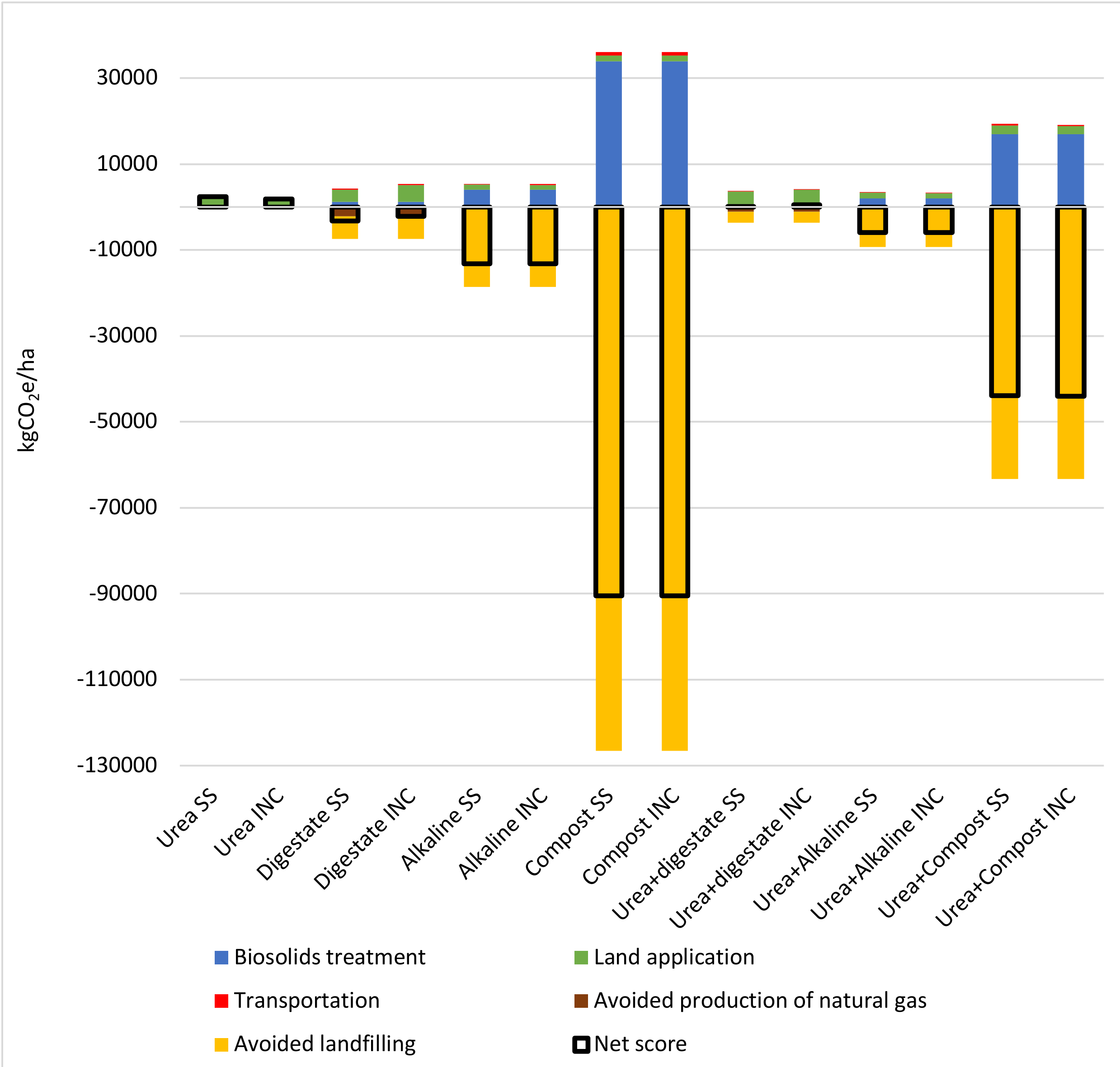

Figure 4.12. Comparative analysis of the life cycle impacts of the 15 different scenarios with the account of the avoided landfilling of sewage sludge

demonstrated the least climate change impact, registering a net negative emission of -90,437 kg-CO_2e. This outcome underscores the importance of the volume of sewage sludge diverted from landfills, especially given that compost production necessitates the largest volume to meet the nitrogen requirements for corn silage cropping. Consequently, this scenario yielded the most substantial climate change credit. This observation accentuates a pivotal facet of LCA: results are profoundly influenced by the defined system boundaries and the study's specific goal and scope.

In summary, it can be interpreted that the outcomes of this comparative analysis are contingent upon the intended disposal method for sewage sludge, be it land application, incineration, or landfilling. Specifically, if the primary intent for sewage sludge is land application, then the scenario with the minimal climate change contribution is the production of urea followed by its land application. In scenarios where sewage sludge would have been incinerated but is instead treated and land-applied (thus diverting it from incineration), the most climate-friendly option is the production and land application of digestate. Finally, if the sewage sludge was originally destined for sanitary landfilling but instead was treated and land-applied, composting the sewage sludge before land application emerges as the most sustainable choice, given its lowest climate change contribution score.

Additionally, there was no notable difference in emissions between the surface-spread and incorporated scenarios. Consequently, this comparative assessment does not favor one field application method over the other due to the negligible differences observed.

Lastly, it's crucial to note that while this carbon footprint assessment treated biogenic CO_2 as neutral in terms of climate change, recent research has begun to challenge this stance. Specifically, studies by Liu et al. in both 2017 and 2019 have questioned the carbon neutrality hypothesis (Liu et al., 2017; Liu et al., 2019). They introduced metric indicators to evaluate the global warming potential of biogenic CO_2, termed GWP_{bio}. These studies emphasize that considering GWP_{bio} can promote biomass utilization and enable a more equitable comparison with fossil fuels.

Chapter 5 Conclusions and recommendations

Municipal biosolids, traditionally classified as waste, represent a valuable resource. As Canada updates its policies and infrastructure, there's a growing need for a clear strategy on how to best use and manage these biosolids. The treatment and land application of biosolids would not only improve wastewater management but also benefit agriculture by offering a rich source of nutrients. These potential benefits align with the principles of the circular economy, turning waste into a resource and completing the nutrient cycle.

The urgency of climate change necessitates immediate and strategic actions. The reduction of greenhouse gas (GHG) emissions is paramount in limiting the rise of global temperatures. In this context, the identification and optimization of biosolids management practices are essential both to minimize emissions and to determine the most efficacious treatment methodologies.

This carbon footprint assessment was conducted to identify the most suitable treatment and application strategies for biosolids within the specific climatic conditions of Canada. A comparative analysis was performed on 15 distinct scenarios, encompassing various combinations of fertilizers, application rates, and methods. The findings revealed that the outcomes of the analysis are dependent on the initial intended use of sewage sludge. If land application was the primary management option for the sewage sludge, urea production followed by land application exhibited the lowest short-term contribution to climate change, with emissions ranging from 1,863.4 to 2,329.9 kg-CO_2e. This was closely followed by the scenario involving digestate biosolids production and land application, primarily attributed to biogas production. Furthermore, the study considered the emissions avoided by diverting sewage sludge from incineration and landfilling. By treating and applying biosolids to land instead of incinerating the sewage sludge, the digestate scenario avoided 262.76 kg-CO_2e, the alkaline stabilization scenario 922.12 kg-CO_2e, and the composted scenario 6283.1 kg-CO_2e. Additionally, when treating and applying biosolids to land instead of landfilling, the avoided emissions were 5289 kg-CO_2e, 18,560 kg-CO_2e, and 126,500 kg-CO_2e for the digestate, alkaline stabilization, and composted scenarios, respectively. The study did not show important differences in emissions due to the biosolids application method.

The study corroborates the idea that treatment of sewage sludge and land application of the resultant biosolids is a beneficial management practice. Despite the emissions associated with biosolids treatment, which may render urea production seemingly favorable for climate change mitigation, the avoided emissions from diverting sewage sludge from landfills alter the results. This underscores the importance of diverting volumes of sewage sludge from landfills as a critical strategy to achieve substantial mitigation of climate change emissions.

The utility of Life Cycle Assessment (LCA) as a pivotal tool in this research underscores its potential in informed decision-making. By offering a holistic perspective on the environmental ramifications associated with diverse biosolids management paradigms, LCA provides valuable insights to policymakers, agronomists, and wastewater treatment stakeholders. The results of this LCA could aid government policy makers to draft and implement new policies that will help guide the practices of farmers and wastewater treatment plant operators. The goal is to provide information to the Canadian government that will help optimize their existing models towards a more accurate accounting of GHG emissions.

For future research recommendations, several avenues warrant exploration:

- Toxicity: Given the potential risks associated with the land-application of biosolids, it's imperative to delve into its toxicity implications, particularly concerning food security and human health. Historically, biosolids have been viewed as waste designated for disposal. As such, comprehensive studies on the safety of biosolids as crop fertilizers are essential. Addressing this will also aid in enhancing public acceptance of biosolids application in agriculture.
- Agronomic benefits: While this study emphasized the nutrient provision aspect of biosolids, a comparative analysis focusing on their agronomic benefits, especially the provision of organic matter to soils, would be insightful. Evaluating the carbon footprint with organic matter provision as the functional unit could offer a different perspective on the environmental impacts.
- Sensitivity and uncertainty analysis: LCA studies are inherently sensitive to their defined goals, scopes, and the data utilized. To reinforce the robustness of such studies, comprehensive sensitivity and uncertainty analyses are recommended. Techniques such as Monte Carlo analysis can be instrumental in understanding the potential range and variability of results based on the distribution of the input data.

Incorporating these elements into subsequent research will not only provide a more comprehensive understanding of the implications of biosolids application but also guide stakeholders in making informed decisions that align with both environmental and public health objectives.

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Chapter 7 Appendices

Tables 7.1, 7.2, and 7.3 were dispatched to three biosolids treatment facilities for preliminary data collection on biosolids treatment processes and their technical specifications. Of these, only Walker Industries responded, furnishing the completed information for their alkaline biosolids production process in Table 7.2. These sheets were designed to streamline the data collection process by presenting information requirements in an easily fillable table format. This approach not only simplified data gathering but also provided insights into the data necessary for a life cycle inventory of diverse biosolids treatment and production methods. The column labeled "estimated values" contains suggested data entries. These are preliminary figures that the respondent can either verify as accurate or incorrect as needed when completing the sheet.

Table 7.1. Data collection sheet on digestate biosolids sent to the Centre de Valorisation des Matières Organiques (Saint-Hyacinthe, Quebec)

Data requirements	Description	Value collected (Please specify unit with your value)	Remarks	Estimated Values
Sludge proportion from total feedstock (sludge + domestic waste+ industrial waste)				
Sludge input capacity	Sludge quantity processed by the treatment plant in dry or wet tons per year			
Digestate output	Digestate produced in dry or wet tons per year			
Yield of digestate	Tons of digestate produced per ton of sludge processed (wet basis)			
Sludge input dry matter content (DM)	Solids content of sludge (kg dry matter per 100 kg wet mass of sludge)			18%-38% Average= 28%

Digestate dry matter content (DM)	Solids content of the final product (kg dry matter per 100 kg wet mass of biosolids)			
Volatile solids in sludge	Volatile solids content in sludge input			0.92 kg VS/ Kg sludge input
Nitrogen in Digestate	Nitrogen content in sludge digestate	Kg-N/ Kg sludge input		
Phosphorus in Digestate	Phosphorus content in sludge digestate	Kg-P/ Kg sludge input		
Potassium in Digestate	Potassium content in digestate	Kg-K/ Kg sludge input		
Methane yield	Methane volume per Kg of volatile solids of the feedstock	Typical sludge yield?		0.52 m3 CH4/ Kg VS
Methane ratio in biogas	Methane content in Biogas before upgrading			50-70% Average= 60%
Hydrogen sulfide ratio in Biogas	Hydrogen sulfide emissions during process			0-1 %
Biogas loss ratio	biogas loss ratio during digestion			5%
	biogas loss ratio during upgrading			2%
	Total biogas loss ratio			7%
Energy, electricity	Electricity for pre-treatment, if any <Data can be obtained in annual or monthly bills of the treatment plant>			- MWh/year
Energy, electricity	Electricity for the digester <Data can be obtained in annual or monthly bills of			- MWh/year

	the treatment plant>			
Energy, electricity	Electricity for the upgrading process <Data can be obtained in annual or monthly bills of the treatment plant>			- MWh/year
	Is there a Combined heat and power (CHP) onsite?	YES OR NO		
Energy, heat	IF YES, give an average value of the overall heat generated onsite, boiler or in CHP			
Energy, heat	Heat for processing plant operations (e.g., in digestion): Energy use <Data can be obtained in annual or monthly bills of the treatment plant>			- GJ/year
Digestate density	Density of the biosolids produced			kg/m3
Distances traveled	Average distance traveled for sludge input, from sludge producing plant to your plant			Default value of 100 km
	Average distance traveled for digestate, from your plant to farm users			Default value of 100 km
Truck capacity class	Capacity of the truck used to			7.5 – 16 tons

	transport sludge/biosolids			
Truck payload	Actual load transported by one truck in ton	Please, specify material and provide %		3.3 tons
Truck fuel consumption	Average fuel consumed per wet ton transported over 100 km			11 kg diesel/ton over 100 km
Water use associated to digestate manufacturing	If any, provide any water consumed, e.g., water used for cleaning purpose			? cubic meter tap water/year
Packaging material, if any	Type of packaging material (e.g., cardboard, polyethylene film, polypropylene container, etc.)			
	Amount per each type (kg packaging material per wet ton of biosolids output)			
Solid waste	If any, please list each type of solid waste generated (and provide amount and unit)			
Wastewater	If relevant (e.g., when there is some water use), the volume of wastewater generated and sent to sewage, in cubic meter			? cubic meter wastewater/year
Other inputs	Please list any other inputs (and provide amount and unit) related to the manufacturing of the biosolids (if not			

	insignificant): e.g., Sodium hydroxide, energy use for loader, etc.			

Table 7.2. Completed data collection sheet for alkaline biosolids sent to N-Viro Systems

Data requirement	**Description**	**Value Collected (Please specify unit with your number)**	**Remarks**	**Our Estimated Values**
Sludge input capacity	Sludge quantity processed by the treatment plant in dry or wet tons per year	31000 wet tons per year		5,700 dry tonnes per year
Lime-stabilized biosolids output	Lime stabilized biosolids produced per year	34000 metric tons per year		39,929 wet tons/year <average of US and Canadian based sludge treatment facilities>
Sludge input dry matter content (DM)	Solids content of sludge (kg dry matter per 100 kg wet mass of sludge) received by N-Viro	22-34%		18%-38% Average= 28%
Lime-stabilized biosolids dry matter content (DM)	Solids content of the final product (kg dry matter per 100 kg wet mass of biosolids)	57.5%-62%		57.5-62%
Yield of biosolids	Tons of lime stabilized biosolids produced per metric tons of sludge processed (wet basis)	factor 1.1 tons of lime-stabilized biosolids produced for every wet ton of biosolids (wet tons)		39,929 / 53,295 = 0.75 wet ton/wet ton

Energy, electricity	Electricity for mixing and for other processing plant operations <Data can be obtained in annual or monthly bills of the treatment plant>	2018: power 72000 kwh (the entire facility), demand 222 kw. 3 silos 3 receiving bins 3 conveyors a mixer and a dryer and facility ventilation (biggest cost= 70% of the total electricity consumption.)		
	OR specify Model of the machinery + Make (quantity produced or processed)			
Energy, heat	Heat for processing plant operations (e.g., in dryer): Energy use <Data can be obtained in annual or monthly bills of the treatment plant>	space heating, natural gas. 2200 Gg joules per month. (400-600 Gg for space heating and 1500 Gg joules per month dryer.)		
	OR specify Model of the machinery + Make (quantity produced or processed)			
Lime-stabilized biosolids density	Density of the biosolids produced	600-800 kg/m3 Average= 700		600-800 kg/m3 Average= 700
Cement kiln dust input	Kg of cement kiln dust per 100 kg of sludge input on a wet or dry weight basis	Dry biosolids require little alkaline. If biosolids are wet, we have to add much more. Average 30-40 kg per 100 kg of sludge		30-40% on a wet weight basis Average= 35%
	Do you have to pay the producer for the cement kiln dust in addition to the transport cost?	YES, or NO?		
	IF YES, provide an average unit price for the kiln dust.	no answer		
Quicklime input	Kg of quicklime per 100 kg of sludge input on a wet or dry weight basis	1.5 %. Coach pure		0-3% Average= 1.5%

Other alkaline material input	Kg per 100 kg of sludge on a wet or dry weight basis	none		0%
	Do you have to pay for that material in addition to the transport cost?	YES, or NO?		
	If YES, provide an average unit price for the material.			
Distances traveled	Average distance traveled for sludge input, from sludge producing plant to your plant	35-40 km hauling from different plants.		
	Average distance traveled for lime stabilized biosolids, from your plant to farm users	10- 100 km average range		
Truck capacity class	Capacity of the truck used to transport sludge/biosolids	32 metric tons big trucks tri-axial trailors (lift and dump material at the back)		7.5 – 16 tons
Truck payload	Actual load transported by one truck in ton	32 metric tons per truck		3.3 tons
Truck fuel consumption	Average fuel consumed per wet ton transported over 100 km	we do not pay for sludge coming or leaving the facility the cost of tranp is zero.		11 kg diesel/ton over 100 km
Water use associated to lime-stabilized biosolids manufacturing	If any, provide any water consumed, e.g., water used for cleaning purpose	110 cubic meters per month most (50%) is to hydrate the biofilter-biofilm (ventilation) in the summer, water is evaporated		? cubic meter tap water/year
Packaging material, if any	Type of packaging material (e.g., cardboard, polyethylene film, polypropylene container, etc.)	none		
	Amount per each type (kg packaging material per wet ton of biosolids output)			
Solid waste	If any, please list each type of solid waste generated (and provide amount and unit)	As part of the process no waste, the only waste is generated is a bit of packaging, domestic waste.		

		Similar to a household		
Wastewater	If relevant (e.g., when there is some water use), the volume of wastewater generated and sent to sewage, in cubic meter	no wastewater		? cubic meter wastewater/year
Other inputs	*Please list any other inputs (and provide amount and unit) related to the manufacturing of the biosolids (if not insignificant): e.g., other materials, energy use for loader, etc.*	*diesel cost: 8000 liters of diesel fuel per year 2 motors and a back-up generator used less than an hour per day. If you are shipping than used more frequently*		

Table 7.3. Data collection sheet for composted biosolids sent to Gaudreau Environnement

Data requirement	**Description**	**Value Collected (Please specify unit with your number)**	**Remarks**	**Estimated value**
What is the ratio (or mass fraction) of municipal biosolids to total feedstock?	Municipal biosolids in kg per 100 kg wet mass of total feedstock			
Composting technique	Aerated Static Pile Composting or Aerated (Turned) Windrow Composting or In-Vessel Composting			<u>Aerated (Turned) Windrow Composting</u>
Woodchips	What is the mass fraction of wood material added to municipal biosolids? (Kg of wood material per 100 kg wet mass of biosolids)			60% wood chips, 40 % biosolids

	What is the source of the woodchips?			
	Do you pay for the product?	YES or NO		
	If YES, please provide average unit price for woodchips			
	What is the mixture of wood?	Please provide the % of wood constituents in the moisture, if available		
	Is it the same mixture throughout the year?	YES, or NO?		
	Are the woodchips processed on-site?	YES or NO		
	IF YES, what kind of machinery is used (e.g., Shredder)	Please specify machinery(is) model + Make (quantity processed)		
Biosolids input capacity	biosolids quantity processed by the treatment plant in dry or wet tons per year			
Compost output	What is the quantity of compost produced in dry or wet tons per year?			12 000 metric tons/ year (composting capacity of Gesterra)
Density of composted biosolids produced	How much does a cubic meter or cubic yard of compost weigh?	kg/m3 or kg/ yd3		
biosolids input dry matter content (DM)	Mass of wet biosolids received at Gesterra (wet tons)/ Total solids/ moisture content			18%-38% Average= 28%
compost dry matter content (DM)	Solids content of the final product (kg dry matter per 100 kg wet mass of biosolids) / Total solids/ moisture content			38.1%
Land use	Surface area of the land used for composting in hectares			
Nitrogen in compost	Nitrogen content in biosolids compost	Kg-N/ Kg biosolids input		
Phosphorus in compost	Phosphorus content in biosolids compost	Kg-P/ Kg biosolids input		

Potassium in compost	Potassium content in compost	Kg-K/ Kg biosolids input		
Mechanical equipment	Please list all the mechanical equipment used during the composting process			biosolids mixers? Loaders, windrow turners, screener, tractors
Energy, electricity/ fuel consumption?	Is there a mechanical mixer for mixing biosolids with wood materials? e.g., MWh/year or Litres of diesel per year etc.. *<Data can be obtained in annual or monthly bills of the treatment plant>*	YES, OR NO?		
IF YES	What is the Electricity/ fuel consumption used for the mixer.?			
	OR specify the Model of the mixer + the make (quantity of biosolids processed by the mixer)			
Fuel consumption	Average fuel consumed by the windrow turner over 100 km.	e.g., litres of diesel per 100 km traveled		
	OR specify the Model of the windrow turner + the make (quantity of biosolids turned by the windrow turner)			
Energy, electricity/ fuel consumption?	Electricity or fuel consumed for compost screening. *<Data can be obtained in annual or monthly bills of the treatment plant>*	e.g., MWh/year or litres of diesel per year		
	OR specify the Model of the screener + the make (quantity of compost screened)			
Distances traveled	Average distance traveled for biosolids input, from biosolids producing plant to your plant			

	Average distance traveled for compost, from your plant to farm users			
Truck capacity class	Capacity of the truck used to transport biosolids/compost			7.5 – 16 tons
Truck payload	Actual load transported by one truck in ton	Please, specify material and provide %		3.3 tons
Truck fuel consumption	Average fuel consumed per wet ton transported over 100 km			11 kg diesel/ton over 100 km
Packaging material if any	Type of packaging material (e.g., cardboard, polyethylene film, polypropylene container, etc.)			
	Amount per each type (kg packaging material per wet ton of biosolids output)			
Solid waste	If any, please list each type of solid waste generated (and provide amount and unit)			
Water use associated to compost manufacturing	If any, provide any water consumed, e.g., water used to wet the compost (to increase compost moisture)?	? cubic meter tap water/year		
Leachate production	Volume of leachate generated in cubic meter if available. *How is the leachate managed on-site? Is the leachate drained and recycled or treated onsite?*	? Cubic meter leachate/ year		
	Is the composting pad made out of clay or asphalt?	YES, or NO?		
	Is the leachate drained and recycled or treated onsite?			
	If the leachate is treated by a wastewater treatment plant on-site, how much energy is used for the treatment?			
Other inputs	Please list any other inputs (and provide amount and unit) related to the manufacturing of the biosolids (if not insignificant): e.g. energy use for loader, etc.			

Tables 7.4, 7.5, and 7.6 below outline the data quality assessment used in modeling the carbon footprint, as detailed in section 3.3.4. Data was evaluated based on two main criteria: reliability and representativeness.

'Reliability' concerns the precise measurement of flows, encompassing material and energy, transportation distances, and release quantities. Conversely, 'representativeness' addresses the geographical and technological precision and breadth of the selected generic data modules (or processes). Furthermore, the potential impact contribution reflects the influence of a specific process or parameter on the results, determined by its average contribution to the examined impact categories. For clarity, a color-coding system has been integrated, as explained in Table 3.3.

Upon assessment, the data was deemed suitable for use, as none of it received a rating of “4”, which would indicate its unsuitability for the case in question.

Table 7.4. Quality assessment of the digested biosolids production and land application data

Life cycle process	Contribution to overall system impact	Quality	
		Reliability	Representativeness
Digested biosolids applied on agricultural land (120 kg-N/ha)	100%		
Anaerobic digestion treatment	31.60%	2	1
Biogas purification	51.50%	2	1
Avoided natural gas consumption	-50.00%	2	2
Land-application	66.9%	1	1
Anaerobic digestion treatment	**100%**		
Construction of digester	1.3%	3	2
Construction of digestate storehouse	0.6%	3	2
Construction of building	0.1%	3	2
Construction of feed tank	0.1%	3	2
Construction of gas holder	0.1%	3	2
Electricity	0.8%	2	2
Organic matter transportation to treatment facility	13.8%	1	1
Sewage sludge transportation to treatment facility	1.1%	3	1
Fugitive methane emissions	82.2%	2	2
Biogas purification	**100%**		
Electricity	7.1%	2	2
Chemical factory construction	6.4%	2	2
Fugitive methane emissions	86.5%	2	2
Land-application	**100%**		
Ammonium nitrate production	12%	2	2
Solid manure loading and spreading	3%	2	2
Combine harvesting	3%	2	2
Fertilizer broadcaster	1%	2	2
Dinitrogen monoxide	78.9%	1	1
Biosolids transportation to farm	2.2%	2	3

Table 7.5. Quality assessment of the alkaline biosolids production and land application data

Life cycle process	Contribution to overall system impact	Quality	
		Reliability	Representativeness
Alkaline-stabilized biosolids applied on agricultural land (120 kg-N/ha)	100%		
Alkaline stabilization treatment	76.60%	2	2
Land-application	23.4%	1	2
Alkaline stabilization treatment	**100%**		
CKD production	44.6%	3	3
Quicklime production	11.1%	2	2
Natural gas consumption	41.6%	1	2
Electricity	1.4%	1	2
Diesel consumption	0.5%	1	2
Sewage sludge transportation to treatment facility	1.3%	2	1
Tap water consumption	0.01%	1	2
Land-application	**100%**		
Ammonium nitrate production	27.0%	2	3
Solid manure loading and spreading	14.6%	2	3
Combine harvesting	7.9%	2	3
Tillage, rotary cultivator	4.2%	2	3
Fertilizer broadcaster	1.3%	2	3
Dinitrogen monoxide	35.5%	1	1
Biosolids transportation to farm	9.5%	2	3

Table 7.6. Quality assessment of the composted biosolids production and land application data

Life cycle process	Contribution to overall system impact	Quality	
		Reliability	Representativeness
Composted biosolids applied on agricultural land (120 kg-N/ha)	100%		
Production/ Treatment of biosolids	95.20%	2	1
Land-application	4.800%	2	1
Production/ Treatment of biosolids	100%		
Machine operation, diesel	10.9%	2	1
Composting infrastructure	1.1%	2	1
Sawdust production	0.001%	2	2
Methane from compost pile	53.1%	3	2
Nitrous oxide from compost pile	33.8%	3	2
Sewage sludge transportation to treatment	1.1%	2	1
Land-application	100%		
Ammonium nitrate production	25.3%	2	3
Combine harvesting	7.4%	2	3
solid manure loading and spreading	46.7%	2	1
Fertilizing by broadcaster	1.2%	2	3
Dinitrogen monoxide	15.4%	1	1
tillage, rotary cultivator	4.0%	2	3
Biosolids transportation to farm	23.3%	2	3