LIFE CYCLE ASSESSMENT OF A NOVEL BIOREFINERY FOR SOLID ORGANIC WASTE IN HALIFAX, NOVA SCOTIA

by

Khashayar Delsoz Bahri

Submitted in partial fulfillment of the requirements
for the degree of Master of Applied Science

at

Dalhousie University
Halifax, Nova Scotia
August 2022

Table of Contents

List of Tables

List of Figures

Abstract

In this study, a novel biorefinery process is proposed for municipal solid organic waste. Source Separated Organics (SSO), Mechanically Separated Organic Fractions (MSOF) of garbage, and dewatered sewage sludge are considered the main waste stream in this project. Anaerobic co-digestion and a novel digestate valorization system make up the suggested biorefinery aiming to maximise energy and nutrient recovery from input waste streams. After digestion of feed streams, generated biogas is sent to combined heat and power (CHP) unit to generate electricity as the main product and heat as a by-product while digestate is sent to a phase separation unit. The solid digestate is sent to composting facilities to further stabilize and sold as the compost product. Liquid digestate is sent to the proposed valorization unit consisting of struvite recovery, ammonia stripping, and algae cultivation. The products of valorization systems are struvite, ammonium sulfate, and algal biomass. Chlorella sorokiniana was the algal species selected for nutrient recovery. The protein content of harvested biomass is a suitable substitute for soybean cultivation as an animal feed.

A life cycle assessment was conducted to compare the environmental impacts of the proposed biorefinery with those of the conventional Halifax regional municipality (HRM) waste management system. HRM waste management implements composting and alkaline stabilization to treat SSO and sludge streams, respectively. Site-specific information was used to compile the inventory data. Additional information from the literature and Ecoinvent database was retrieved when necessary. Seven different impact categories were used to compare novel biorefinery and HRM scenarios, including fossil resource scarcity, freshwater eutrophication, global warming potential, land use, marine eutrophication, terrestrial acidification, and water use. OpenLCA software was employed to carry out the impact assessment stage of the LCA and perform further analysis.

The results of the study showed a dominant contribution in environmental saving associated with electricity generation from the CHP plant in all evaluated impacts except land use. After electricity generation, ammonium sulfate, and struvite products showed higher contributions in reducing the environmental burdens of the waste management system. Produced algae (which substituted soybean cultivation) did not show a significant impact except for the marine eutrophication category.

List of Abbreviations

ICI	Industrial, Commercial, and Institutional
WSF	Waste Stabilization Facility
RDF	Residual Disposal Facility
FEP	Front End Processing
MSOF	Mechanically Sorted Organic Fraction
OFMSW	Organic Fraction of Municipal Solid Waste
MSW	Municipal Solid Waste
MBT	Mechanical Biological Treatment
OLR	Organic Loading Rate
FW	Food Waste
TAN	total ammonia nitrogen
FAN	free ammonia nitrogen
LFG	Landfill Gas
FRS	Fossil Resource Scarcity

Acknowledgments

Foremost, I would like to express my sincere gratitude to my supervisors, Dr. Azadeh Kermanshahi Pour and Dr. Matthew Eckelman for their support and patience during this study. Besides my advisors, I would like to thank ASPIRE CREATE and Dr. Rob Jamieson for providing financial support during my study. Finally, I would like to thank my family for their support during all steps of my education.

Chapter 1: Introduction

1.1 Problem Statement

Population growth has led to increased solid waste generation, and waste management has emerged as a pressing global challenge. By 2050, the Municipal Solid Waste (MSW) generated annually in the world is predicted to increase from 2.01 billion tonnes in 2016 to 3.40 billion tonnes (Kaza et al., 2018). In Canada, the total amount of collected solid waste increased by 3.5 million tonnes per year from 2002 to 2016 (Environment Canada, 2018). The waste management issue becomes more substantial in Canada, as the generated waste per capita is higher compared to other developed countries. For instance, Canada generated 965 kg solid waste/capita/year in 2010, while this number was 735 and 500 kg/capita/year for United States and average of European countries, respectively (Richter et al., 2017).

MSW consists of different streams from a wide range of sources including residential, commercial, institutional, construction and demolition, municipal services, industrial plants, and agricultural (Tchobanoglous & Kreith, 2002). Notably, the term "solid" does not indicate that the stream solely exists in a solid condition; rather, solid waste can also refer to some streams that are in a liquid or semi-liquid state (US EPA, 2020). The definition of MSW and its sources vary in different literature and legislations. In the most general form, MSW is defined as durable goods, nondurable goods, containers & packaging, food wastes, and yard trimmings generated in residential (e.g. single and multi-family houses), commercial (e.g. office buildings, restaurants, etc.), institutional (e.g. schools, libraries, hospitals, etc.), and industrial (e.g. administrative and packaging, *not* process waste) sectors (US EPA, 1999b). Another source of MSW is treated sewage sludge, also known as biosolids, which is a waste stream generated after treating municipal wastewater (Roy et al., 2011). Landfilling of biosolids, like other kinds of solid waste, may have a negative influence on the environment due to leachate and greenhouse gas production (H. Wang et al., 2008).

Mismanagement of MSW contributes to a wide range of concerns, including adverse environmental and health impacts. These impacts are arising in different pathways. For instance, methane emission from landfills, which results from the anaerobic decomposition of organic waste, is among the air pollutants that significantly contribute to climate change and global warming (Du et al., 2017). Methane pollutant from landfills is estimated to account for 3-19% of the anthropogenic methane sources globally (S. Kumar et al., 2004). Despite the air emissions, landfill leachate is another pathway that instigates environmental deterioration. Leachate is generally formed because of passing water from waste and dissolving soluble compounds through chemical and biological processes (El-Fadel et al., 1997). Leachate includes a variety of pollutants, including inorganic contaminants (i.e., nitrogenous, and heavy metals), organic contaminants (carbonic and organic acids), and biological contaminants. These pollutants potentially introduced contamination into adjacent groundwater, surface water and soil (Mukherjee et al., 2015).

In response to the mentioned challenges, Nova Scotia (NS) province has implemented waste management approaches since 1995 by initiating Solid Waste Resource Management (SWRM) strategies. The first legislated act was the "Environmental Act" (Nova Scotia Environment, 1994), which emphasized the importance of strategies such as reducing waste generation, reusing, and recycling. As a result of this legislation, NS was one of the first provinces that achieved the national target of 50% waste diversion by 2000 (Giroux, 2014). Environmental Goals and Sustainable Prosperity Act (EGSPA) (Nova Scotia Environment, 2007) was another amendment that was set to reduce the amount of waste produced by each individual annually from 430 kg waste/person/year in 2007 to 300 kg waste/person/year by 2015 (NS Enviro, 2009).

1.2 Objectives

According to the lessons learned from waste management practices, improving a complex system like waste management calls for a combination of economic, environmental, and social factors. Each treatment/disposal process involved can be optimized to improve the waste management system and bring it closer to sustainability (Asefi et al., 2020). HRM is implementing a combination of composting for source-separated organics, alkaline stabilization for sludge, and sanitary landfill disposal facility with an organic stabilization unit for managing the generated waste (Divert NS, 2018; HRM, 2011). Although the HRM waste management system is already implementing source separation, and separate treatment for diverted organic waste, no energy recovery is currently practiced. Energy production from waste is relatively cleaner compared to fossil fuels, except for natural gas (Khan et al., 2022). Waste-to-energy technologies can also decrease dependency on fossil fuels as well as reduce the land requirement for waste disposal (Khan et al., 2022; Noor et al., 2013).

In this study, a novel biorefinery is proposed for organic solid waste in Halifax Regional Municipality (HRM) in Nova Scotia province, Canada. The proposed system aims to include energy production by implementing anaerobic digestion while enhancing nutrient recovery from digestate. A life cycle assessment (LCA) was conducted to measure the environmental performance of the proposed system and compare it with that of the conventional waste management scenario in HRM. LCA results were used to examine the proposed system with respect to its environmental burden and identify future research directions to further decrease the environmental burden of the proposed system.

1.3 Document Organization

After the introduction chapter, Chapter 2 of the document provides a literature review of related waste management technologies, digestate valorization technologies and life cycle assessment background information. Chapter 3 investigates the main stages associated to a life cycle assessment study including the scope of the study, life cycle inventory analysis, and life cycle impact assessment. In life cycle inventory analysis subsections, process description and mass flow analysis associated with each facility are provided. The inventory data presented in Chapter 3 is used to develop models in openLCA software to generate the results of the environmental impact. Results generated for various environmental impact categories along with sensitivity analysis are presented in Chapter 4. Conclusions and proposed future work and limitations are discussed in Chapter 5.

Chapter 2: Literature Review

In this section, the scientific background of systems involved in this study is provided. Initially, a short process description and process parameters are provided for each waste management technology associated with this study. After that, an introduction to the digestate valorization system, which is the novelty of the proposed biorefinery in this study, is provided. Finally, an introduction to LCA methodology and background information on LCA in waste management is provided.

2.1 Waste Management Technologies

2.1.1 Anaerobic Digestion

Anaerobic Digestion (AD) and composting are identified as the most promising biological processes to treat solid organic waste globally. Although composting technology has been implemented for a longer time, an increasing interest in AD systems can be seen in recent years. Depending on the operational conditions and microbial community, the main product of AD systems is methane, hydrogen, volatile fatty acids, and lactic acids. Among these main products, only methane is common and fully commercialized (Monlau et al., 2015).

AD reaction phases can be summarized into four main steps: 1) in the **hydrolysis** phase, the insoluble organics and large molecules are hydrolyzed to soluble organics, 2) in the second phase, the **acidogenic** bacteria will further digest soluble organics to volatile fatty acids (VFAs) along with NH3, CO2 and H2S, 3) utilizing products of the second phase, the **acetogens** will produce acetic acid, CO2 and H2 in the third phase, and 4) finally in the last steps, methane is produced by **methanogens** by either catalyzing break down of acetate or reaction of CO2 and H2 (Christensen, 2011; Khanal, 2011). Process parameters, including pH/alkalinity, temperature, and retention time, play a critical role in controlling this chain of reactions and increasing the digester efficiency.

The reactor PH range mostly affects the performance of the methanogenesis step. While methanogens activity is optimum at the pH range of 6.5 to 7.2, other steps have a wider operational pH range that makes them less sensitive (e.g., the acidogenesis step works in the 4.0 to 8.5 pH range). In AD, the pH range is controlled by alkalinity with the major alkalinity components including carbonate, bicarbonate, and hydroxide ions. Alkalinity provides a buffer capacity for AD to resist perturbations in pH (Appels et al., 2008; Ward et al., 2008).

Temperature impacts different aspects of AD including physicochemical properties of media, microbial growth rate, metabolism, partial pressure of H2, etc. Anaerobic reactions can take place in different ranges including psychrophilic (14 to 23°C), mesophilic (30 to 40°C), or thermophilic (50 to 60°C). Both endergonic and exergonic reactions take place in AD which has made choosing the proper temperature more challenging. Higher temperatures will increase the solubility of organics which increases overall degradability that leads to higher organic reduction and pathogen removal. However, it has its own drawbacks such as increasing ammonia concentration, which is inhibitory for anaerobic reactions (Appels et al., 2008; Mao et al., 2015).

Retention time in anaerobic digestion is divided into two separate variables; solid retention time (SRT) refers to the average time that solid particles are in the digester while hydraulic retention time (HRT) addresses the average time that the liquid phase spends in the digester. A balance should be achieved to reach proper degradation with optimum retention time to avoid larger digesters (Khanal, 2011). Short retention time will lead to methanogens washout and incomplete components breakdowns (especially hard-to-digest materials such as lipids), which increase VFA concentration in the effluent. A favorable retention time has stable concentrations in output. It should be noted that a portion of microorganisms exits the reactor as solids in effluent, as a result, cell growth should be adjusted with HRT and SRT as well (Appels et al., 2008).

Organic Loading Rate (OLR) is a parameter that indicates the organic content that is being fed to the reactor. One of the most crucial process variables in AD is achieving proper OLR that the system can handle (Igoni et al., 2008; Khanal, 2011). OLR is defined by feed organic contents flowrate per unit volume of the digester (de Lemos Chernicharo, 2007):

$$OLR = \frac{S \times Q_{in}}{V_R}$$

Where S is influent organic matter concentration (i.e., kg VS/ m^3, kg COD/ m^3 or kg BOD/ m^3), Q_{in} is influent flowrate (m^3/d), and V_R is reactor volume. OLR should be optimized based on the feedstock (amount of biodegradable content in the influent) and type of the reactor (Khanal, 2011). Basically, lower organic loads lead to less available organics for the methanogens which yield to lower biogas production. On the other hand, higher organic loads change the balance between acidogenesis and methanogenesis reactions (Igoni et al., 2008). This alteration produces more acids in the reactor, resulting in decreasing pH, which affects methanogens more than other species (C. Liu et al., 2017; Ostrem & Themelis, 2004). As a result, each system has an optimum OLR(D.-H. Kim & Oh, 2011a).

AD systems are divided into two categories based on their capability of handling organic load: low-rate and high-rate digesters (Figure 2-1). low-rate digesters operate with an organic loading rate in the range of 1-2 kg COD/m^3 day Anaerobic ponds and septic tanks are among the commonly used low-rate digesters. Containers need to be washed out after each set of digestion to remove accumulated solids, energy loss and odor emissions are expected. In addition, low-rate digesters require higher volume, increasing the capital cost. Therefore, these systems are not suitable for high quantities of influent (Khanal, 2011; Metcalf & Eddy & Tchobanoglous, 2014).

Unlike low-rate digesters, high-rate digesters are employed to treat higher organic rates (higher than 5 kg COD/m3 day). Various configurations of anaerobic digesters are developed to enable achieving higher organic loading rate, this can be achieved by decoupling HRT and SRT as solids require more retention time since they need to be dissolved first. Four common methods to optimize HRT and SRT to achieve higher retention times include (1) attached growth system to form biofilms that can keep biomass for a longer time in the reactor. Media such as plastic, gravel, sand and activated carbon are used as biomass carriers; (2) in the second approach, microorganisms form granules and flocs which enhance retention of the biomass in the reactor; (3) Biomass recycling aims to recirculate formed microorganism flocs to increase retention time in the reactor; and (4) membrane bioreactor retains biomass in the reactor, which leads to higher average retention time (Khanal, 2011; Metcalf & Eddy & Tchobanoglous, 2014).

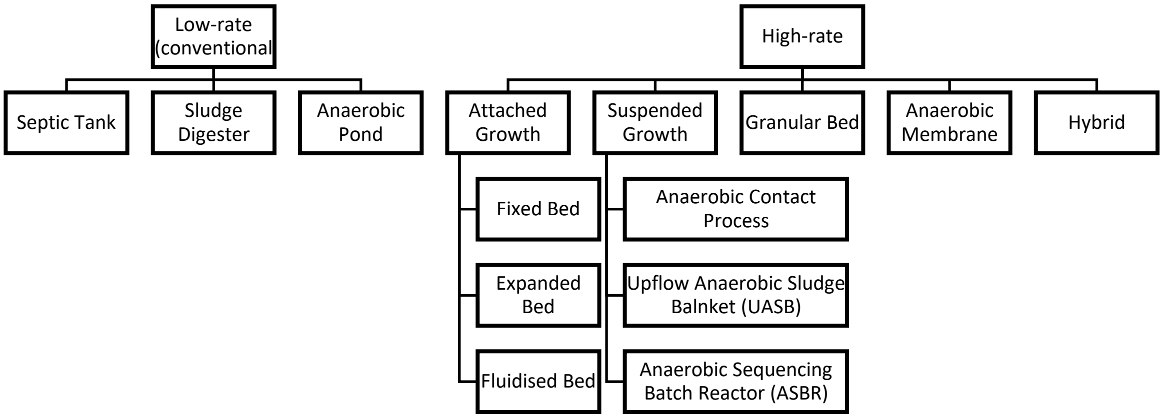

Figure 2-1 Summary of different anaerobic digestion systems based on organic loading rate

2.1.2 Composting

The composting process is one of the oldest technologies that have been used in the waste treatment industry. Despite a long history of composting in agriculture, commercial facilities for municipal and industrial waste were not established until the middle of the 20th century (Stoffella J & Kahn, 2001). Composting refers to the process of aerobic digestion of solid organic waste that leads to the formation of new components and heat (Christensen, 2011). Although this process naturally occurs to digest organic residues by living organisms, this section focuses on controlled composting technologies to treat solid organic waste.

Reactions involved in composting can be described in three main phases. Since temperature is the main factor that affects the kinetic rates (degradation rate), these phases are explained by their temperature conditions. The first phase is performed in mesophilic culture and decomposes easily degradable materials. As the temperature gradually rises, it enhances degradation rates for the rest of the biomass. The second phase starts when the temperature reaches up to 70 °C, which is at the thermophilic range. This high temperature is a direct consequence of the degradation reactions which are exothermic. Finally, the third phase involves the mineralization of less biodegradable components while humidifying lignocellulosic materials. The final step can take weeks to be completed depending on the characteristics of the feedstock (Christensen, 2011; Debertoldi et al., 1983).

Composting process is directly impacted by the composition and flowrate of the input Regarding the degradation rate of different waste sources, Gutiérrez et al. obtained reaction rates of different waste types (Figure 2-2). Reaction rates, which translate to the degradability of different waste streams, were estimated by using biological oxygen demand (BOD) while keeping Oxygen availability and temperature constant. In similar conditions, higher oxygen demand simply means more biodegradable biomass. The results of their study showed a high degradation rate for the waste streams that contained an organic fraction of MSW (Gutiérrez et al., 2017).

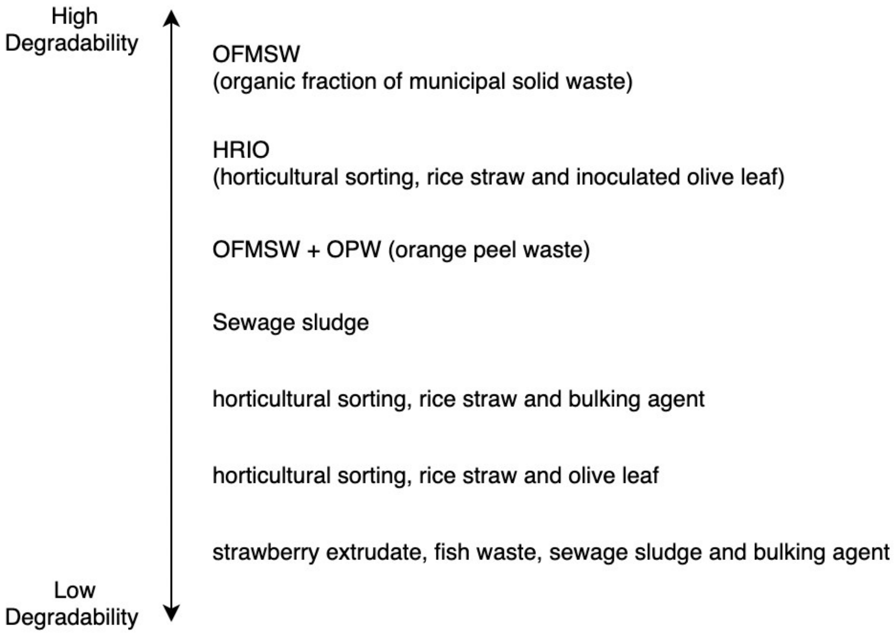

Figure 2-2 Degradability of different waste types based on BOD

The product, commonly referred to as compost, is usually used as fertilizer. Compost can provide various organic matters and nutrients to the soil while enhancing soil cation exchangeability and texture (US EPA, 2002). The composting process can be divided into three main steps: pre-treatment, main reactor where the composting reaction occurs, and post treatment (Cerda et al., 2018; Christensen, 2011). A common process diagram and factors that should be considered in each section are summarized in Figure 2-3 (Cerda et al., 2018; Christensen, 2011).

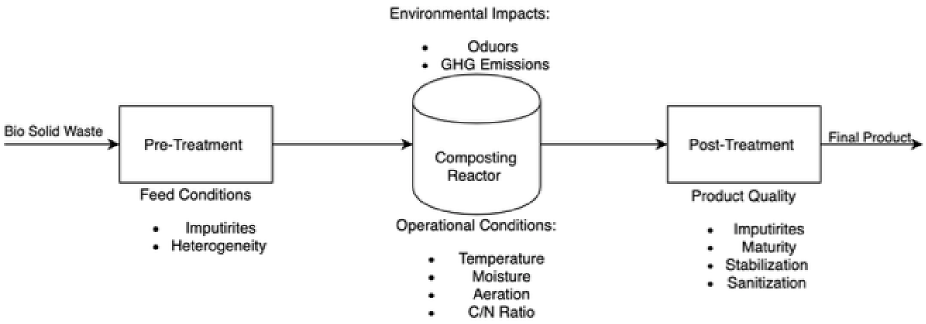

Figure 2-3 Summary of important factors in composting technologies, regenerated based on information from (Cerda et al., 2018; Christensen, 2011)

The most widely implemented technologies in full-scale composting plants can be split into two primary categories: open composting and enclosed composting. Windrow composting is one of the most common technologies for the open category. In these systems, aeration will naturally take place and no heat is provided for the facility. For the enclosed systems, channel composting and reactor-based composting is more common. In closed systems, proper aeration must be provided, thus better control of the aeration system (such as exhaust air recirculation, temperature and conditioning of air) should be provided (Christensen, 2011).

Temperature is another important process parameter that highly affects the performance of the system (Hamoda et al., 1998). Figure 2-4 is representing the temperature profile during different stages of the composting process. Click or tap here to enter text. Temperature profile can be divided into main three stages: rising temperature, stationary phase, and cooling phase. In a controlled composting process, the stationary phase is controlled at a lower temperature to avoid nutrient loss (Ajmal et al., 2020). A balance between temperature and retention time should be achieved (higher temperature will decrease the retention time). Moreover, the temperature can affect the sanitization and stabilization level of the treated stream which is important legislation-wise (Christensen, 2011; Petric & Selimbašić, 2008).

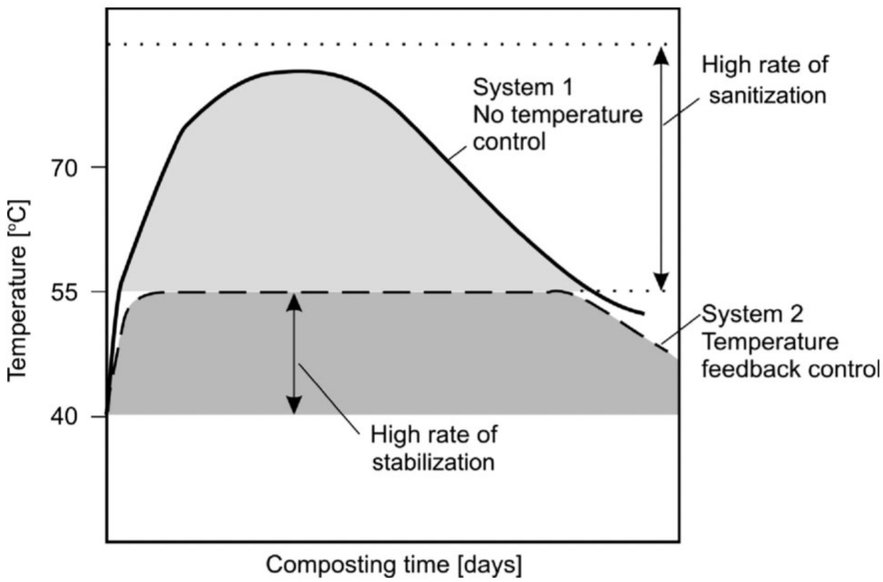

Figure 2-4 General temperature profile of natural and controlled composting process (Christensen, 2011)

Moisture Content is another important parameter in composting reactors. Water is a vital element in composting as it provides the wet working environment that most microorganisms need for moving gases and liquids through their cell membranes. During composting the heat generated evaporates large quantities of water from the composting mass. Moisture content above 35–40% is recommended to prevent composting inhibition due to lack of moisture (Christensen, 2011). Oxygen availability or aeration can be divided into four zones. For each zone, provided oxygen should be higher than the calculated stoichiometric value to ensure proper distribution of air (Christensen, 2011). Mentioned zones are as follows:

- Zone 1: high oxygen consumption rate as the pile temperature increases (3 days)
- Zone 2: Peak demand (5 to 10 days)
- Zone 3: Cooling (the more readily degradable materials have now been used)
- Zone 4: Stabilization is completed, and maturation/humidification begins.

2.1.3 Alkaline Stabilization

Wastewater treatment facilities generally have two main outputs: treated (clean) water and separated solids which are commonly referred to as sewage or waste sludge (hereinafter referred to as sludge). The sources of solid effluent can vary based on the wastewater treatment facility process. Primary sludge (sludge from the primary settling tank), activated sludge (sludge from the secondary clarifier, after aeration tank), and aerobically or anaerobically digested sludge are among the most common sludge waste streams (Tchobanoglus et al., 2003). Biosolid is referred to as treated sewage sludge. It is identified that the biosolids and sewage sludge terms are used interchangeably in governmental and scientific documents. However, in this study, the difference between these terms is considered and biosolid is only referred to as “treated” sludge. Generated biosolid can be potentially further treated to produce value-added products as it contains organic matter and nutrients (GIROVIČ, 1996). Among suggested applications, land application is one the most attractive options as it substitutes fertilizer and improves the soil quality (Fisher et al., 2019).

Treating sludge before the land application is required to mainly reduce pathogens (viruses, bacteria, etc.) and vector (birds, insects, rodents, etc.) attractiveness. Considering these two indicators, the processed biosolid is categorized into Class A and B. To apply treated biosolid on land, meeting Class A is required. Class A standard requires biosolid to have less than 3 most probable number (MPN) per 4 g dry total solids for salmonella sp., less than 1 Plaque Forming Unit (PFU) per 4 g dry total solids for enteric viruses, and less than 1 viable helminth per 4 g dry solids for viable helminth ova (US EPA, 2003). To achieve this goal, different treatment systems including alkaline stabilization, anaerobic digestion, and composting are commonly used (Tchobanoglus et al., 2003). This section focuses on the alkaline stabilization process which is the implemented system in HRM for biosolid.

Figure 2-5 shows an overall schematic of the alkaline stabilization process. In this process, high pH conditions (above 12) are achieved by adding an alkaline admixture which results in microbial reaction elimination. In addition, viruses, bacteria and other microorganisms (i.e. pathogenic organisms) will be inactivated (Tchobanoglus et al., 2003; L. K. Wang et al., 2007). While lime is the most used alkaline admixture, other industrial by-products such as cement kiln dust, lime kiln dust, limestone, alkaline fly ash, and wood ash can also be used in the alkaline stabilization process (Logan & Harrison, 1995).

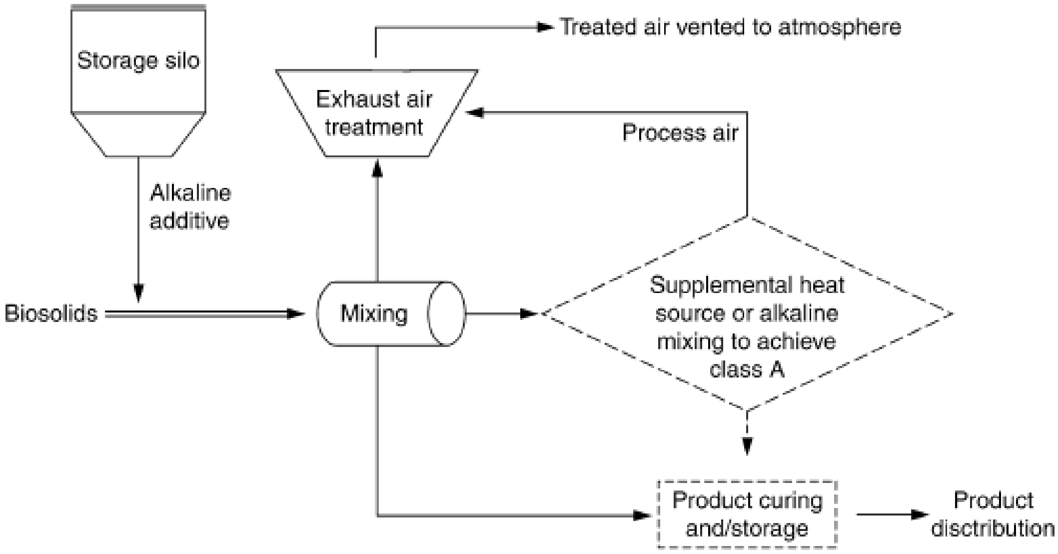

Figure 2-5 Schematic of alkaline stabilization process. Adopted from (US EPA, 2000).

Although the details of chemical reactions in alkaline stabilization is not fully known (L. K. Wang et al., 2007), following chemical reactions are proposed in case lime is used as alkaline admixture (Tchobanoglus et al., 2003):

$Ca^{2+} + 2HC_3^- + CaO \rightarrow 2CaCO_3 + H_2O$

$2PO_4^{3-} + 6H^+ + 3CaO \rightarrow Ca(PO_4)_2 + 3H_2O$

$CO_2 + CaO \rightarrow CaCO_3$

$RCOOH + CaO \rightarrow RCOOCaOH$

$Fat + Ca(OH)_2 \rightarrow glycerol + fatty\ acids$

High temperature is another contributing factor in alkaline stabilization. According to the US EPA fact sheet, above pH of 12 and 52°C temperature should be maintained for 72 and 12 hours, respectively, to achieve a Class A product (US EPA, 2002). Heat is generated when the alkaline admixture is mixed with dewatered sludge. Main exothermic reactions are hydration of calcium oxide and solid organic matter degradation (Rodríguez et al., 2012; Valderrama, Granados, & Cortina, 2013):`

$Hydration\ of\ quicklime: CaO + H_2O \leftrightarrow 2Ca(OH)_2$

$Degradation\ of\ organic\ matter: C_6H_{15}O_6\mathrm{N}_{(S)} + 6O_2 \rightarrow 6CO_2 + 6H_2O + NH_3$

Mentioned reactions can cause the temperature to rise above 50°C. Good mixing is critical in the mixing reactor to ensure sufficient contact between lime and sludge (Tchobanoglus et al., 2003). Additional heat to increase the temperature to above 70°C

(Class A requirement) is either supplied by adding additional lime or using another heat source (e.g. electricity and natural gas) (US EPA, 2002; L. K. Wang et al., 2007).

2.2 Digestate Valorization

Regardless of AD system type, its digestate potentially is rich in vital components such as nitrogen, phosphorus and potassium. In addition, it includes non-treated organic carbons that require further stabilization (Guilayn et al., 2020; Monlau et al., 2015). Considering circular bio-economy concepts, successful waste management can be achieved by integrating different waste treatment technologies (Wainaina et al., 2020). A potential pathway for AD systems is to use generated digestate to produce a variety of by-products. Even though creating value-added products from digestate might improve the overall performance of the AD systems from an economic and environmental standpoint (Sheets et al., 2015). This section aims to review the commercial technologies that can recover precious value-added materials from anaerobically digested streams by utilizing supplementary units that can be coupled with the AD process.

2.2.1 Digestate Characteristics

Whole digestate application for land use is one of the most common and well-established methods for digestate final use (Fuchs & Drosg, 2013). Generally, the nutrient concentration is increased in an AD process due to the decomposition of organic material. For instance, a higher ammonium (NH4+-N) concentration is observed in digestate compared to the initial digester feed due to breakdown of large organic molecules (Möller & Müller, 2012). However, for agriculture applications, not only the nutrient concentration is important but also the accessibility of nutrients for plants is critical. This accessibility is increased in the AD process as a result of nutrient mineralization and releasing them from complex compounds to available N, P, Ca, and Mg (Guilayn et al., 2020).

However, there are some limitations associated with the direct application of digestate for land use. The concentration of hard-to-digestate components (e.g. lignocellulosic material) is generally increased as the majority of degradable components are converted to biogas (Guilayn et al., 2020). Other limitations include the existence of heavy metals and pathogens. Levels of pathogens in digestate highly depend on the AD operational conditions (especially temperature) and the influent's pathogen concentration. Lagoons and other stabilization, and valorization methods can be helpful to prevent pathogens from entering the soil. In addition, direct land application of raw digestate can cause volatile emissions and nutrient runoff as the balance of nutrients required for soil should be taken into consideration (Fuchs & Drosg, 2013; Sheets et al., 2015). For instance, it is reported that in Europe the raw digestate dry matters does not meet the soil improver standards and it does not contain enough nutrients to be considered as fertilizer (Bamelis et al., 2015; Monlau et al., 2015; Styles et al., 2018). To this aim, it is recommended that raw digestate be upgraded so that it may be used in agricultural applications by using digestate valorization procedures. Implemented digestate valorization methods are expected to (Guilayn et al., 2020):

- Eliminate/decrease undesired characteristics of digestate (i.e. chemical precipitation, nitrification/denitrification)
- Optimize transport cost and emission by concentrating the valuable components (i.e. reducing volume by solid/liquid separation)
- Produce value-added products which meets common standards and have greater and wider market acceptance
- Ensure controlled and homogeneous production of fertilizer for land application

2.2.2 Valorization Systems

The target of this section is to evaluate different processes that upgrade the digestate to be implemented as fertilizer or soil improver. Nutrient accessibility, uniformity in nutrient content, odor nuisances, ability to improve soil structure, salinity, pathogens, and potential application with currently available technology and pricing are the primary aims and issues to accomplish (Guilayn et al., 2020). Produced fertilizers must meet the requirements of standard fertilizers. Among these requirements, the concentration of N, P, and K can be mentioned. P and K usually can be found as K2O and P2O5 in the product while all types of N are considered suitable N for plant uptake. An example of the required concentrations for each fertilizer is provided in the table below (van Eekert et al., 2012):

Table 2-1 Required concentration for each fertilizer, retrieved from (van Eekert et al., 2012)

Fertilizer	Requirement
N-fertilizer	N-content > 15%

NK-fertilizer	N+K_2O > 15%
P-fertilizer	N+P_2O_5 > 18%
PK-fertilizer	K_2O+P_2O_5 > 18%
NPK-fertilizer	N+K_2O+P_2O_5 > 15%

Phase-separation

To separate a liquid fraction and a solid fraction, the produced digestate passes through phase separation. This phase separation can be achieved by using common equipment such as screw press. The liquid fraction is intended to concentrate nutrients, whereas the solid fraction mostly comprises organic molecules (Fuchs & Drosg, 2013). After phase separation, different nutrients might end up in different streams (Figure 2-6). Nitrogen, for instance, is mostly transferred to liquid streams mainly because it is dissolved as ammonia. On the other hand, phosphorus forms solid particles and can be mostly found in the solid stream. Different nutrient destination depends highly on the type of implemented separation process (Askri et al., 2016; Guilayn et al., 2019).

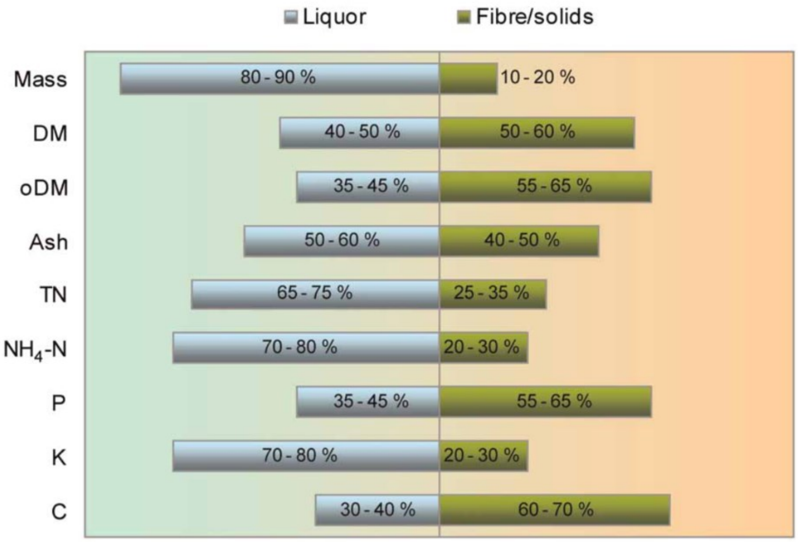

Figure 2-6 Distribution of nutrients after phase separation. DM: dry matter, oDM: organic dry matter, retrieved from (Fuchs & Drosg, 2013)

Ammonia Stripping

The main mechanism of this system is to convert dissolved ammonia to free ammonia in the gaseous phase. This step is commonly performed by chemical air stripping, non-chemical air stripping, vacuum stripping, and steam stripping. Figure 2-7 shows the schematic of air and steam stripping. Higher pH and temperature facilitate the mass transfer of dissolved ammonia to gaseous form. NaOH or $Ca(OH)_2$ are commonly injected into the stream to increase the pH before the stripper. The dissolved ammonia transfers to the gaseous phase in the stripping column by injecting liquid and gas streams, usually counter-currently. In the air stripping technique, the gaseous phase from the stripper is sent to an adsorber to be washed with acid. The cleaned air can be recycled to the stripper. In the steam stripping method, the gaseous phase is sent to a condenser to extract ammonia solution (Lorick et al., 2020; Palakodeti et al., 2021; van Eekert et al., 2012).

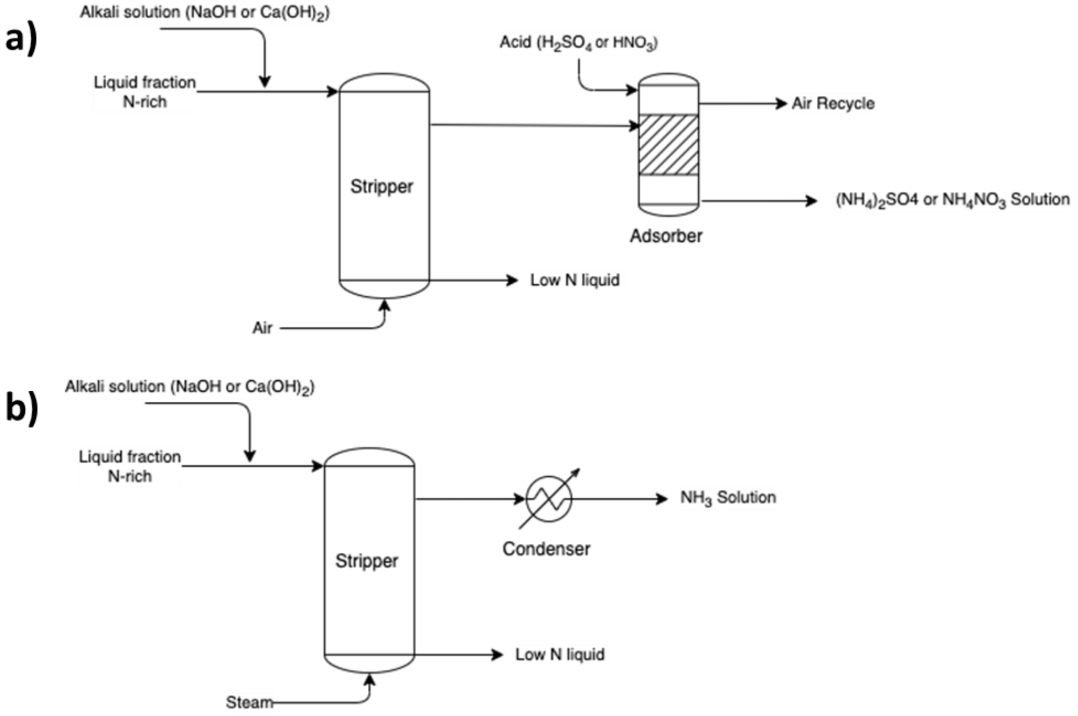

Figure 2-7 Ammonia stripping system schematic. Figure a is ammonia stripping with air while figure b is showing steam stripping, regenerated using information from (Lorick et al., 2020; Palakodeti et al., 2021; van Eekert et al., 2012).

Ammonium sulfate or ammonium nitrate are the end products of this process, with ammonium sulfate being more prevalent. The efficiency of this process, which is defined as the percentage of removed nitrogen from the fed digestate, is reported near 100% theoretically while the practical efficiencies are lower in the 80-90% range to decrease the operational costs by operating the unit in a lower pH or temperature. The concentration of recovered ammonium sulfate is in the range of 25-40% (H. Wu & Vaneeckhaute, 2022). Moreover, in some full-scale designs, low-quality gypsum (such as synthetic gypsum) is used in the adsorber to produce CaCO3 in parallel (Bauermeister et al., 2009).

The amount of NaOH and sulphuric acid depends on the nitrogen levels and system operational values. A range of 2-2.5 kg of NaOH and 3.5-7 kg of sulfuric acid per kg N in the feed are required (van Eekert et al., 2012). In addition to chemical requirements, electricity and heat (especially in steam stripping) are required. Having in mind all of the chemicals and energy requirements, N-stripping systems cost is about 4.5-8.6 €/m3 digestate (H. Wu & Vaneeckhaute, 2022) or 1.9-3.2 €/kg N removed (van Eekert et al., 2012).

Struvite Recovery

Chemical precipitation/crystallization techniques have been commercially implemented for P, Ca^{2+}, Mg^{2+} and ammonium recovery (Egle et al., 2015). Other recovery methods such as electrochemical, ion exchange separation and biomineralization are not fully developed and are currently under evaluation on a laboratory scale (Kataki et al., 2016). P is commonly recovered as Struvite ($MgNH_4PO_4.6H_2O$). In struvite precipitation, usually precipitation takes place in a mixing reactor with an alkaline condition. The desired alkalinity can be achieved by adding chemicals such as (NaOH, MgO, NH_3, etc.) or CO_2 stripping (Tao et al., 2016). In addition to pH adjustment, additional magnesium sources are commonly provided by adding magnesium salts to the reactor. By adding additional Mg, P will become the limiting reactant, which increases the P recovery. $MgCl_2$, $MgSO_4$ and MgO are among the common components to provide the required Mg for precipitation. A recycling stream is considered at the outlet of the reactor to enhance the precipitation efficiency as some fine struvite might escape without recovery (Crutchik & Garrido, 2011; Kataki et al., 2016).

The following reaction takes place in the reactor to form struvite (Corona et al., 2020). The required Mg:N:P molar ratio is 1:1:1 for struvite precipitation, however, a higher Mg:P ratio (i.e. 1.5:1) is required to initiate the precipitation process (Egle et al., 2015):

$$Mg^{2+} + NH_4^{+} + PO_4^{3-} + 6H_2O \rightarrow MgNH_4PO_4 \cdot 6H_2O$$

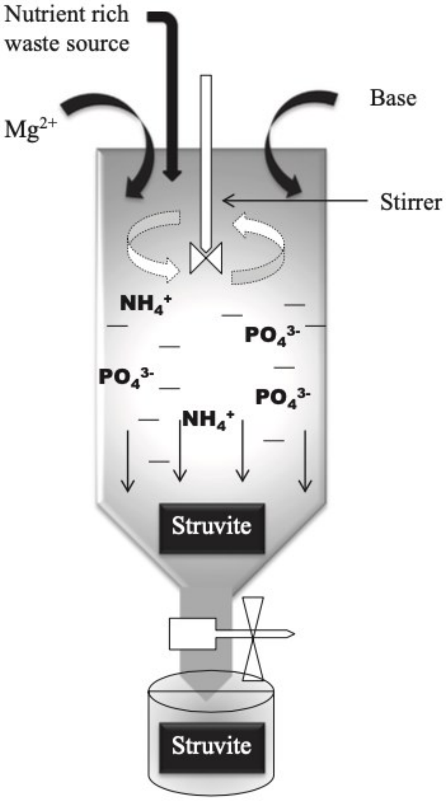

Figure 2-8 Chemical precipitation reactor for struvite recovery, retrieved from (Kataki et al., 2016)

Struvite recovery can capture 80-90% of P, but it is not considered a nitrogen recovery method as it only captures 10-40% of dissolved ammonia. The produced struvite not only can be sold as a by-product but also decreases operating and disposal costs due to reducing the potential risk of system plugging because of struvite precipitation and reducing the volume of digestate, respectively (Kataki et al., 2016; H. Wu & Vaneeckhaute, 2022).

AirPrex® is one of the first employed commercial processes for struvite recovery (Figure 2-9). In this process, an aeration tank is used before precipitation. Aeration leads to CO2 stripping that increases the pH up to 8 (Egle et al., 2015). Two main mechanisms operate in both high and low pH regions that can explain this pH increase. In lower pH values, bicarbonate is converted to CO_2 and water by aeration, which consumes H^+. In higher pH ranges, aeration converts bicarbonate to carbonic acid. Produced carbonic acid is then converted to CO_2. In both mechanisms, pH is increased as a result of aeration and equilibrium will be achieved via the following reactions (Cohen & Kirchmann, 2004):

Low pH: $HCO_3^- + H^+ \rightarrow CO_2 + H_2O$

High pH: $HCO_3^- + H_2O \rightarrow H_2CO_3 + OH^-$

$H_2CO_3 \rightarrow CO_2 + H_2O$

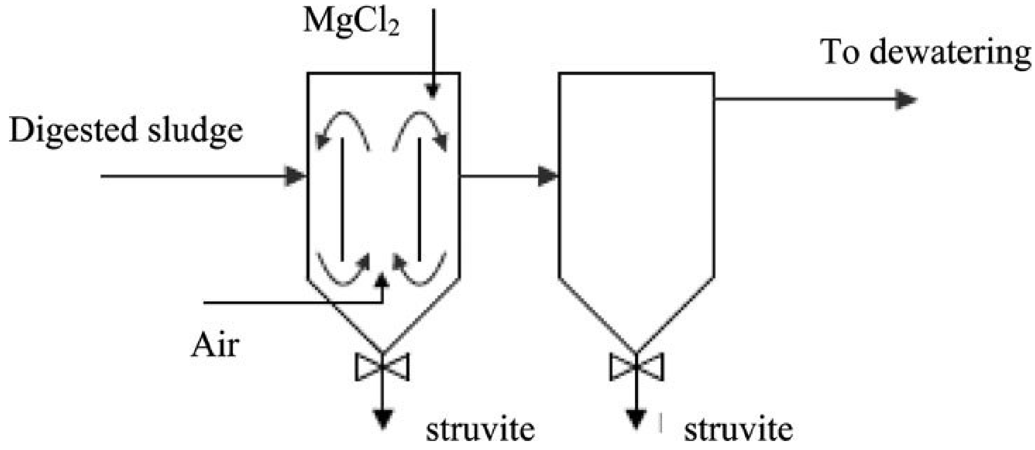

Figure 2-9 Overview of AirPrex process (Desmidt et al., 2015)

In addition to pH adjustment, aeration maintains the internal flow of crystals in the reactor (internal recycling of crystals) which yields a larger crystal size (Desmidt et al., 2015). Precipitant (usually $MgCl_2$) is added to the aeration tank to fix Mg:P molar ratio at 1.5:1 (Egle et al., 2015). After the first tank, finer crystals are settled in the second reactor. Struvite is cautiously harvested from the bottom of both tanks (Desmidt et al., 2015). The HRT of the reactor is 8 h (Nieminen, 2010).

Evaporation

Evaporation further reduces the volume and concentrates the nutrients of the liquid fraction of the digestate. The outputs of the evaporator are thickened concentrate (about 50% dry matter) and water ammonium (Figure 2-10). The concentrate output can be mixed with the solid fraction of digestate or be directly applied to the land. The water-ammonium mixture can be further treated by an ammonia scrubber, ammonia stripping coupled with a scrubber, or reverse osmosis. Another commonly used technology in the thermal treatment of digestate is vacuum evaporation which required lower energy (Barampouti et al., 2020; Guilayn et al., 2020; Vondra et al., 2019).

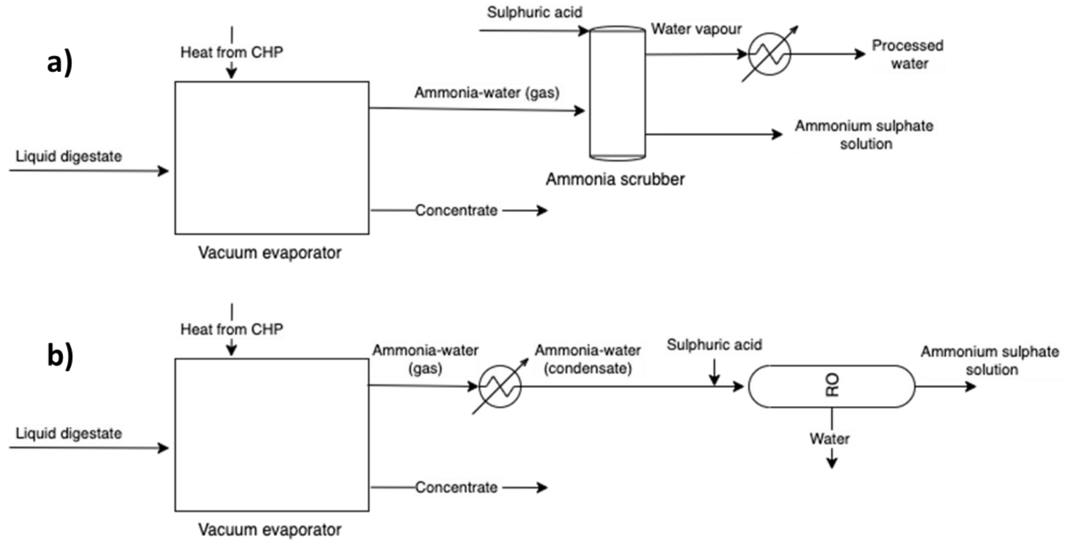

Figure 2-10 Liquid fraction valorization via evaporation method. Figures a and b are evaporators coupled with scrubber and reverse osmosis (RO), respectively.

Sulphuric acid wash is used in both systems, which reacts with ammonia in the gaseous phase and produces ammonium sulfate solution. In the scrubber scenario, the reaction is exothermic, as a result, most of the water remains in the gas form which will be condensed afterward. In the RO process, a liquid ammonia-water solution is sent to the RO. The final water stream from both processes is safe to be discharged into the natural environment or to be used as recycled water in the plant if it is possible (Vondra et al., 2019).

Regarding energy and chemical consumption, it should be mentioned that both of provided systems are commercially developed at full scale. For instance, a biogas plant, Pont Langlois in Normandy France, is currently using evaporation coupled with reverse osmosis to treat the liquid digestate. The required heat for evaporation is supplied by hot water from the CHP unit to maintain the evaporator temperature around 38 °C while the electricity consumption is 0.025 kWh/m3 of digestate. About the mass balance, 26.3 t digestate is produced daily. The liquid fraction rate is about 25 m3/day. The concentrate stream (7.1 t/d) with 20% dry matter is mixed with solid digestate and is used as a soil amendment. The RO produces 2t/d ammonium sulfate and 20t/d clean water which is disposed of in a nearby river. The efficiency of nitrogen capturing is close to 98% (Bamelis et al., 2015). It is reported that evaporation can reduce up to 50% of the liquid fraction volume which can decrease the transportation cost. Evaporation is more feasible when enough heat is produced by biogas combined heat and power generation unit, due to the high heat requirements of the system (Fuchs & Drosg, 2013).

The main operational limitations and problems are stripping columns scaling, corrosion, and low total solids requirements for input (Fuchs & Drosg, 2013). Another limitation is the low concentration of nitrogen in the product, about 6-10% wet weight, which is lower than commercial fertilizers (Guilayn et al., 2020; Sheets et al., 2015). Other bottlenecks of this method are operational problems (like plugging and foaming), and high energy requirements (efficiency will decrease in low temperatures) (Fuchs & Drosg, 2013; Sheets et al., 2015).

Membrane filtration:

Membrane technology produces a higher standard product for land application compared to not-treated liquid digestate by concentrating nutrients in the liquid stream. In addition, a processed water stream is produced which can be safely disposed of or reused (Gienau et al., 2018; Guilayn et al., 2020). Figure 2-11 indicated commercial membrane technologies implemented in full-scale plants. The liquid fraction after the separator is sent to a decanter that further separates the solids to prevent membrane fouling. The solid output of the decanter is mixed with the solid fraction of digestate. In the decanter, flocculants such as mineral-polymeric flocculants may be added to increase the separation efficiency. The addition of flocculants in large-scale plants showed effective results as they increased the removal of solids by 84–93% while releasing just 4% of nitrogen into the solid streams (Zarebska et al., 2015).

After the decanter, the liquid stream is sent to Ultra-filtration (UF) membrane. The UF membrane with a pore diameter in the range of 0.001-0.1 µm is usually used. P exists in streams as fine particles, hence UF showed a better P removal efficiency compared to N. UF retentate can be recycled to the system or mixed with the liquid fertilizer. Finally, the reverse osmosis (RO) membrane with pores smaller than 0.001 µm can be coupled with UF to capture 95-99% of the remaining dissolved ammonia salts (Piaia, 2013).

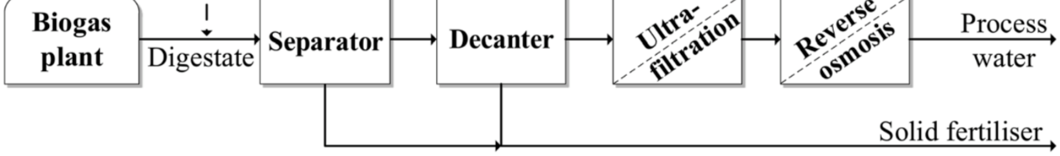

Figure 2-11 schematics of a membrane technology used for treating liquid fraction of digestate, extracted from (Gienau et al., 2018).

The main limitation of the membrane technology for digestate valorization is its high operational requirements which increase the cost and decrease the marketability. The total energy consumption of the processes reported around 20-30 kWh per m3 of digestate which is high compared to other technologies. Around 50-70% of this energy consumption is for the ultra-filtration step since it requires high flow rates and pressures. In addition to energy, potential fouling issue is another limitation of this technology (Gienau et al., 2018).

Algae Cultivation:

Micro-algae cultivation using AD digestate has shown promising potential to recover nutrients and produce valuable biomass. In addition to available nutrients in digestate that can be used by algae species, CO2 in biogas can be used as a carbon source making coupling AD with algae cultivation more economical (Bjornsson et al., 2013; Nguyen et al., 2019). Different applications can be considered for the harvested biomass. One option is to convert the biomass to biodiesel via acid transesterification or base esterification by converting fatty acids to esters (Chinnasamy et al., 2010). Other potential products from microalgae are biomethane, bio-hydrogen, animal feed, value-added products for pharma drugs, etc. (Enamala et al., 2018)

Various factors determine the algae growth rate and the composition of produced biomass including light, carbon, and nutrient sources (especially nitrogen and phosphorus), temperature, pH, and O2 removal/CO2 uptake (Enamala et al., 2018). The proper range of mentioned factors highly depends on the type of microalgae specie. The most utilized species for digestate-fed media are *Chorella sorokiniana*, *Chlorella vulgaris*, *Chorella pyrenoidosa*, *Chorella minutissima*, *Scenedesmus obliquus*, *Scenedesmus bijuga*, and *Scenedesmus accuminatus* (Enamala et al., 2018; Iyovo et al., 2010; Sayedin et al., 2020; Xu et al., 2015).

As mentioned previously, N and P are the essential nutrients required for algae growth. Various N:P ratio is suggested in literature; (Cai et al., 2013) suggested that the optimum N:P ratio for C. vulgaris is 7 while the optimum N:P ratio for Scenedesmus to grow without limitation is approximately 30. (Stiles et al., 2018) has suggested that 7:1 would be a proper ratio generally for algae growth.

Ammonia inhibition is one of the challenges of algae cultivation that has been identified as a potential bottleneck that might restrict algae growth on digestate. The tolerance of algae to ammonia directly depends on the type of specie used in the cultivation process (Monlau et al., 2015). For instance, C. sorokiniana can tolerate ammonia concentrations up to 3500 mg/l (Q. Wang et al., 2019). In addition to ammonia, high turbidity can adversely impact algae cultivation as well. Both of these limitations can be addressed by diluting the digestate (Sayedin et al., 2020).

2.3 Life Cycle Assessment (LCA)

2.3.1 LCA Basics

In this section, a brief introduction to the LCA method is provided. Life Cycle Assessment (LCA) has been suggested to evaluate the environmental performance of different systems in 3 main scales, based on the European LCA handbook: Micro-level improvements (improvement and comparison of products), Meso-macro level (policies development), and Accounting (monitoring products and policies) (EU Commission, 2010). As defined by ISO 14040, Life Cycle Assessment (LCA) is defined as the "compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle" (ISO, 2006). ISO summarizes the LCA approach in four different steps:

1. Goal and Scope Definition (LCI)
2. Inventory analysis phase (LCI)
3. Impact assessment phase (LCIA)
4. Interpretation phase

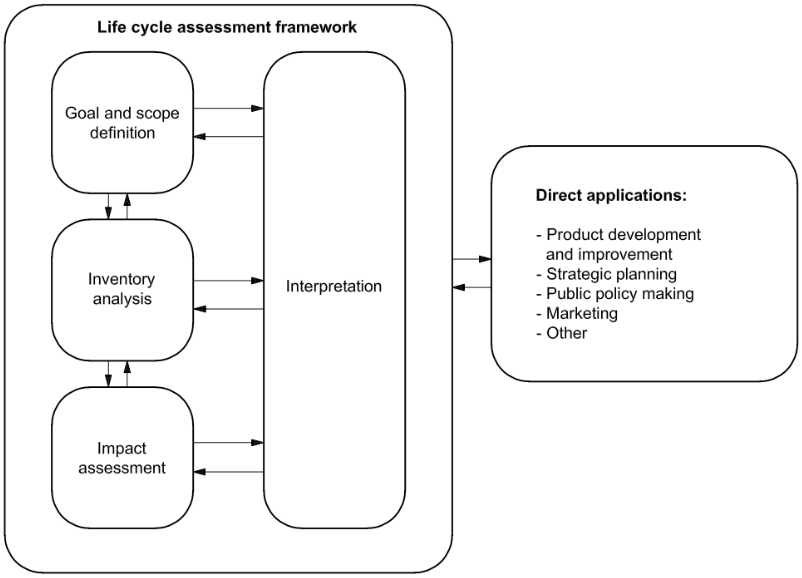

Figure 2-12 Infographic of different LCA stages and their relation (ISO, 2006)

Goal and scope definition

The first stage describes the system's general characteristics, such as the objective and scope of the study. The goals of an LCA range from identifying appropriate technology for a region (e.g. evaluating alternative waste management methods) to a narrower focus (e.g. evaluating the performance of a newly added unit to a process). In addition, general concepts of the study including services, the functional unit, and system boundaries are defined (Allen & Shonnard, 2012; ISO, 2006). The system boundary is a "set of criteria specifying which unit processes are part of a product system" as defined by ISO. A balance between available databases and desired system scope should be achieved; a wider boundary simply will lead to a more developed model and more accurate results regarding the environmental burdens of an action. However, achieving an appropriate database can be complex in many cases (Allen & Shonnard, 2012).

Functional Unit is defined as the "quantified performance of a product system for use as reference unit" in the ISO. In defining the functional unit, concentration is on the function of the products instead of their physical properties. Attributional Life Cycle Assessment (ALCA) and Consequential Life Cycle Assessment (CLCA) are two methodologies that can be practiced in different systems, depending on their specifications. In attributional LCA physical inputs and outputs are considered as they occur. On the other hand, in Consequential LCA modeling, the consequences of the processes on the background system are measured instead of their physical effect (EU Commission, 2010; Finnveden et al., 2009).

Life Cycle Inventory (LCI) Analysis

In the second phase, a list of inputs and outputs is created. This is step is one of the iterative sections in the study; during data collection and calculation, LCI might require some alterations and improvements to get closer to the study goal (ISO, 2006). A common challenge in LCA is how to address the multi-functionality of the processes. This issue arises when the discussed process has multiple inputs/outputs or an open-looping system (for instance, the effluent of anaerobic digestion, which can be considered as a waste for the system or can be used in nutrient recovery systems) (EU Commission, 2010; Finnveden et al., 2009). Different solutions have been provided to address multi-functionality in an LCA. In the following, some of the most common and standard procedures are explained by their priority to use as defined by guidelines and literature (EU Commission, 2010; ISO, 2006; Schrijvers et al., 2016):

i. Subdivision of the processes:

Subdivision can solve the multifunctionality by converting the system into multiple single-function units in a black box process unit (a system unit that includes multiple systems inside it). In addition, the subdivision of the systems can generally help the accuracy and review procedure of the case too (EU Commission, 2010).

ii. System expansion / substitution:

If subdivision could not be implemented, the next option is to use system expansion, relying on expanding the system boundaries to cover connected sub-processes. Practically, system expansions are not always the most optimized way to treat multifunctionality since other added systems can be multifunctional as well. In addition to that issue, two major problems should be answered too: a) the quality of the products in the alternative path and b) the suitable alternative path (which marginal process should be chosen) (EU Commission, 2010; Finnveden, 1999).

iii. Allocation:

According to ISO standard, using allocation should be avoided as much as possible. However, there are situations that using allocation is inevitable. The allocation method is simply splitting the inputs or products based on their weight, economic value, energy, etc. (EU Commission, 2010; Finnveden et al., 2009).

Life cycle impact analysis and interpretation

In the third step, LCA impact analysis, results of LCI are used to analyze the significant environmental impacts of the process. To evaluate these effects, environmental impact categories and category indicators are employed (ISO, 2006). This step can include a wide variety of impacts such as natural resource use (water and energy consumption), environmental impacts, and socio-economic impacts (Allen & Shonnard, 2012). The choice of impact category is subjective, as a result, no framework or regulation is defined, but it should reflect a comprehensive set of environmental issues.

The final step, interpretation, presents outcomes of LCI and impact categories together. This section is a report-based document to deliver the conclusion or recommendation of the study to decision-makers. Another important matter that is hidden in the concept of an LCA study, is the iterative approach of it (EU Commission, 2010; ISO, 2006). The iteration can be started once the initial goal is set, and initial results are achieved. The following figure is showing a schematic of an ideal iterative approach (EU Commission, 2010). In each iteration, avoiding misleading and key parameters should be checked.

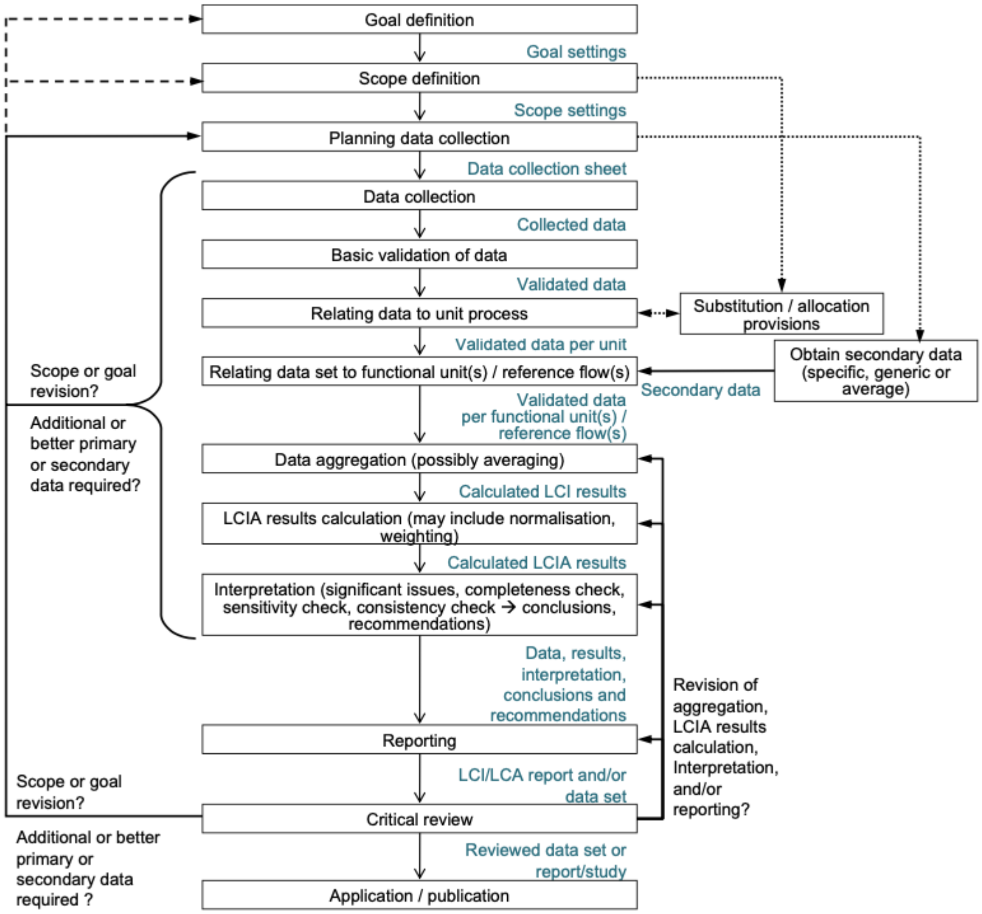

Figure 2-13: Details of an iterative approach defined by European commission handbook (EU Commission, 2010)

2.3.2 LCA and waste management

LCA has been widely used in previous studies to assess the environmental impact of different waste management systems. Among the benefits of LCA in waste management, the following items can be mentioned (Allesch & Brunner, 2014; Fatta & Mol, 2003; Karmperis et al., 2013).

- Evaluating impact of different material and energy consumption due to the selective burden of the whole system,
- Assessing long-term environmental impacts of the system, and
- The ability to sync the result with other analysis methods such as life cycle cost (LCC) and techno-economic evaluation

Liu and Rajagopal investigated the application of incineration, anaerobic digestion (AD), and gasification as waste-to-energy technologies in the United States for different waste streams including agriculture residues, animal manure, forest residues, and MSW (B. Liu & Rajagopal, 2019). Mentioned technologies were compared by net energy gain, potential renewable energy production, and greenhouse gas (GHG) emissions impact categories. It should be noted that WtE technologies can be considered a renewable way to produce energy. As a result, although in some cases the system might consume large amounts of energy from that specific state's energy mix (which most of it is not renewable) it produces a larger amount of renewable energy (positive net energy). As a result, both net energy gain and consumed energy in the system should be evaluated to thoroughly compare different systems. For municipal solid waste (MSW), incineration offered the highest net energy gain while anaerobic digestion had lower emitted CO_2 equivalent due to lower energy consumption. It is worthwhile to mention that the digestate impacts are not considered in the system boundary. As a result, other environmental impacts of AD (such as avoiding chemicals and energy due to implementing digestate) are not considered.

In another study by Tan et al., the environmental impacts of landfill gas recovery, incineration, AD, and gasification for MSW in Malaysia were evaluated (Tan et al., 2015). They identified that incineration, AD, and gasification had higher environmental performance compared to landfill gas recovery from a carbon credit point of view. In respect to energy production, incineration had the highest energy production followed by AD and gasification while gasification showed the highest potential for carbon mitigation followed by AD and incineration. Authors also recognized that other factors such as moisture of the waste (which can affect incineration adversely), waste availability (AD can also use agricultural), and marketability of potential renewable fuel from biogas should be considered, which were not included in their scope of the study.

Jensen et al. compared the waste management systems of Germany and Denmark for organic municipal waste (Jensen et al., 2016). Two main different approaches including a combination of composting and anaerobic digestion (German waste management system), and incineration (Danish waste management system) were considered. They identified that incineration had better performance in marine eutrophication and acidification compared to composting and AD systems. This was due to identified nutrient leakage by using compost on land, ammonia emissions, and potential methane and CO_2 leakage from the composting/AD processes. On the other hand, AD and composting scenario showed a better performance in photochemical ozone formation, terrestrial and freshwater eutrophication. This difference was due to the amount NO_x, NH_3, and SO_2 emissions from incineration and phosphorus recovery in AD.

In another similar study, Bernstad and la Cour Jansen compared incineration, decentralized composting and centralized anaerobic digestion for food and organic waste in Sweden. The digestate application on land was also considered for AD technology. In addition, different end-products of AD biogas including biogas upgrade to light vehicles fuel and generation of heat and electricity by CHP were considered. In general, the AD scenario with the biogas upgrade had the best overall environmental performance among all other scenarios. For GWP, incineration showed the least favorable performance mainly due to direct emissions followed by composting and AD. Composting scenario had the highest adverse impact on acidification due to nutrient enrichment.

In general, based on studied literature, AD is proposing a great potential to reduce the waste management system impact compared to the composting process by generating energy while using the digestate as a fertilizer product. A controversial result is observed comparing incineration and AD LCA studies, based on the location and the scope of the study. Regarding the Global Warming Potential (GWP), direct emissions from the processing facility and avoided emissions by producing energy were the most important factors. The contribution of the incineration plant is widely changing from one case to another one. Although the incineration produced more energy than AD in all cases, its related emissions such as NOx and sulfides were larger than AD. Moreover, biogas production provides more flexibility and a wider range of end-products (i.e., light fuel, electricity and heat, etc.) compared to incineration products which are basically heat and electricity.

Chapter 3: Life Cycle Assessment

3.1 Goal and Scope

The main objective of this study is to evaluate and compare the environmental performance of a proposed biorefinery and a conventional waste management system for the treatment of the organic fraction of municipal solid waste.

The system boundaries of the study include different stages of waste management from collecting and treating to the endpoint (grave). The endpoint covers the disposal of material or their specific application (e.g. land application). Different sub-systems that are considered are collection and transportation of the waste to facilities, different treatment facilities (composting, biosolid processing facility etc.), disposal facilities, distribution of the product and application of the product. Five different environmental impact categories are selected for this evaluation: global warming potential, cumulative energy demand, ozone layer depletion, acidification potential, and eutrophication potential.

Two main scenarios are evaluated in the LCA study; (1) the conventional waste management system (currently implemented system) which is the baseline case of the LCA, and (2) the proposed novel biorefinery. The current system is using composting, alkaline stabilization and landfilling as the main foreground systems. The proposed scenario mainly uses the biorefinery as well as the landfilling and composting systems. For both scenarios, the background systems, including wastewater treatment for leachate treatment, electricity mix, diesel production, and air pollution control is connected to the foreground systems.

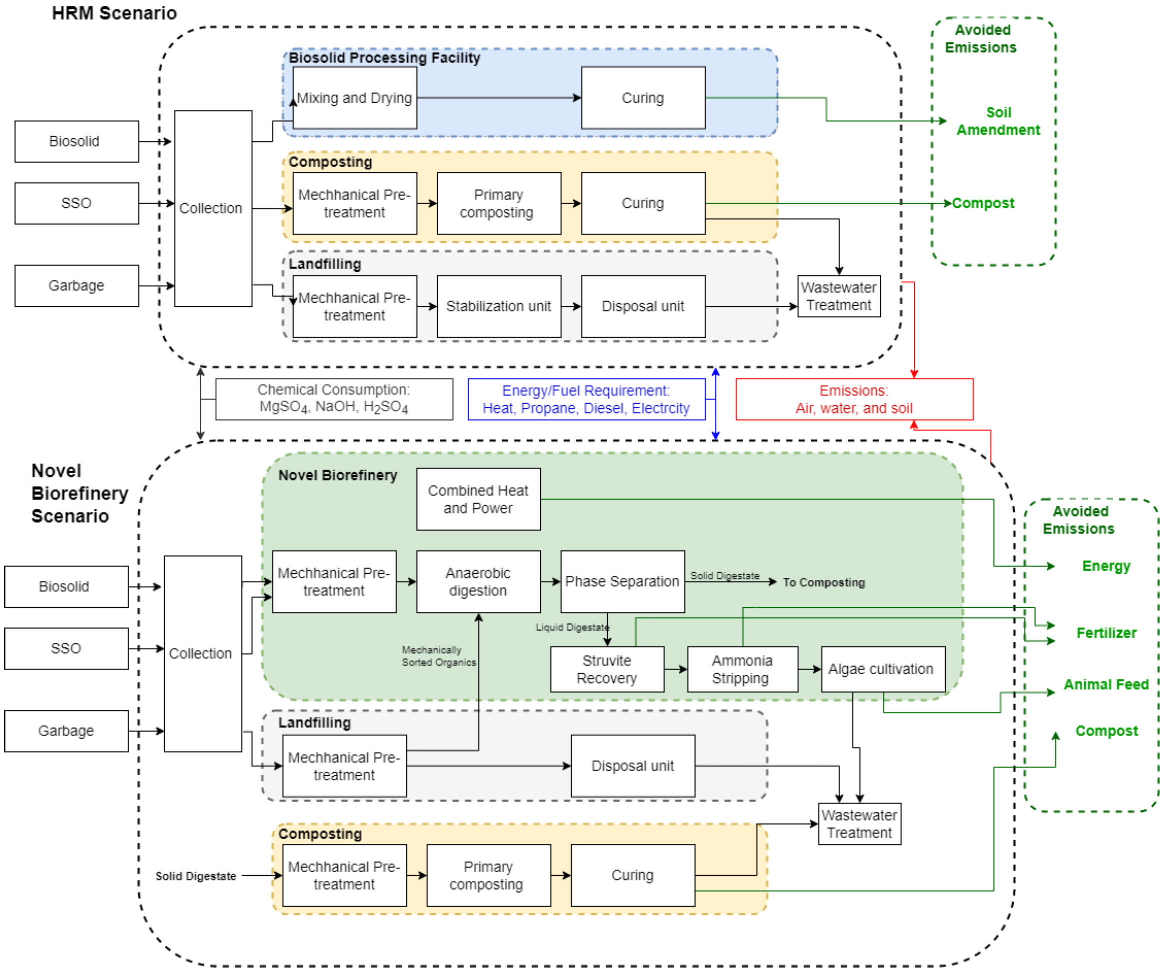

Figure 3-1 The system boundary of the waste management system in two scenarios. Different sub-systems are proposed for each scenario including composting, alkaline stabilization, wastewater treatment (for leachates), landfilling, collection, and transportation, novel biorefinery.

3.1.1 Geographical Scope

Nova Scotia solid waste management is divided into 7 main regions (Figure **3-2**). Region 4, Halifax Regional Municipality (HRM), has the highest population of these regions, resulting in the highest waste generation. The geographical scope of this study is the solid waste generated and treated within the border of HRM.

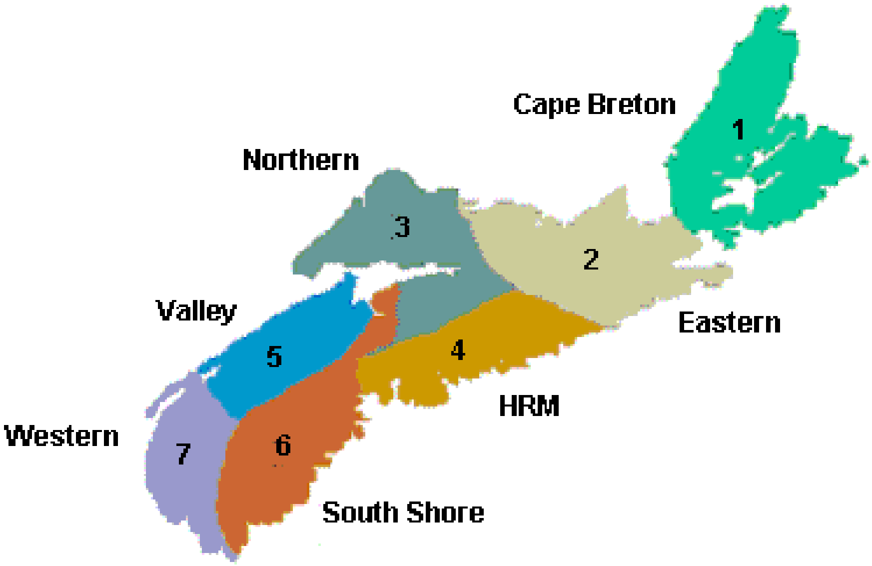

Figure 3-2 Nova Scotia solid waste management regions. The geographical scope of this study is the waste generated and treated within the Halifax Regional Municipality (HRM) region.

3.1.2 Input Waste Streams

Three solid waste streams containing organic fractions are considered within the scope of the study: source-separated organics (SSO), garbage, and dewatered sludge from Wastewater Treatment Facilities (WWTFs). Table 3-1 is showing available historical data on the amount of mentioned waste streams in Halifax Regional Municipality (HRM). A summary of each waste stream is provided below:

- Source Separated Organics: The SSO is separated organics from both residential and industrial, commercial, and institutional (ICI) sections, collected from curbside in green carts. This stream mainly consists of food waste, and leaf and yard wastes(HRM, 2020a). According to NS government databases, 35,000 t separated organics (35% of the total curbside collection) were collected from the residentials (NS Government, 2020b). In addition, 16,000 t source-separated organics were collected from the ICI section in 2019 (NS Government, 2020a).
- Garbage: Garbage is the mixed waste stream that remains after recyclables and organics have been separated, and it is collected from both residential and ICI locations. Total amount of collected garbage from residentials was 45,000 t in 2019 (NS Government, 2020b). Generated garbage from ICI section is not included in this study as it is sent to external landfill facilities and is not treated within the HRM boundaries.

- Sludge: According to communication with Halifax Water, 30,000 t of partially dewatered sludge (28.5% TS) is generated in HRM per year. Currently, dewatering takes place in the WWTF and dewatered sludge is transferred to the biosolid processing facility by truck (Halifax Water, 2020c).

3.1.3 Functional Unit

The Functional unit (FU) is set as total amount of municipal solid waste generated in HRM in one year. Considering 2019 as the reference year, FU consists of 51,000 t SSO (35,000 and 16,000 t from residentials and ICI respectively), 45,000 t garbage (only residential) and 30,000 t partially dewatered sludge.

Table 3-1 HRM (Region 4) Available solid waste information on residential and ICI sections. Empty cells are indicating no data was available. All values are rounded.

Waste (Origin)	2015 (t/y)	2016 (t/y)	2017 (t/y)	2018 (t/y)	2019 (t/y)	Source
Garbage (Residential)	58000	50000	45000	-	45000	(NS Government, 2020b)
Organics (Residential)	35000	36000	34000	36000	35000	(NS Government, 2020b, 2020a)
Recyclables (Residential)	17000	19000	19000	-	18000	(NS Government, 2020b)
Organics (ICI)	-	17000	16000	16000	16000	(NS Government, 2020a)
Landfilled (Residential)	-	47000	42000	-	40000	(NS Government, 2020c)
Landfilled (ICI)	77000	81000	83000	-	90000	(NS Government, 2020c)

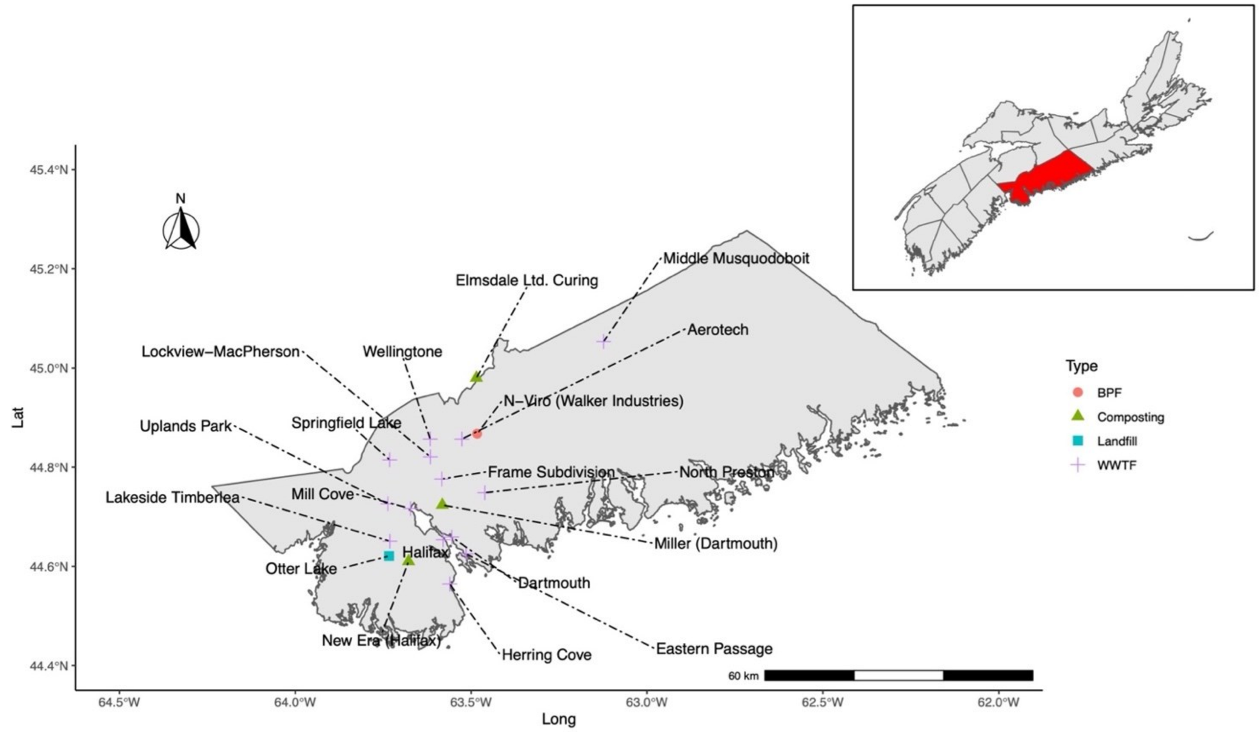

Figure 3-3 Map of Halifax Regional Municipality solid waste region and related facilities. The SS-MSW is collected by truck and is sent to composting facilities to be treated. The sludge from Wastewater Treatment Facilities (WWTF) is sent to Biosolid Processing Facility (BPF). The garbage is collected and sent to Otter Lake Landfill facility.

3.2 Life Cycle Inventory Analysis: Current System

3.2.1 Composting

Composting technology is implemented in HRM to treat collected source-separated organics (SSO) from residentials and ICI sections. Currently, two central facilities are responsible for composting treatment: Ragged Lake Facility (Halifax) and Miller Facility (Dartmouth) (NS Government, 2020d). The process flow diagram of these facilities is presented in Figure 3-4. Current composting process includes 3 main steps: (1) front-end sorting, (2) decomposition step (primary composting and curing), and (3) back-end sorting. The gaseous and leachate output of both facilities are managed by biofilter and leachate collection systems respectively (Stantec, 2013).

Process Description

After receiving the SSO in receiving hall, a belt conveyor passes the waste to the front-end sorting and grinding station where front-end rejects (C2) are separated. C2 stream includes non-compostable materials and metals, which are removed by staff (manually) and magnets. The non-compostable stream is sent to the landfill while the separated recyclables are sent to material recovery facility. Both streams are not included within the scope of this study as they do not contain any organic fraction. After sorting, sorted feed is sent to an 8" grinder and shredder with 2.5-inch screen size in Ragged Lake and Miller facilities respectively. This step aims to adjust the particle sizes to enhance the degradation process (Aim Environmental Group, 2017; HRM, 2016). Reduced particle sizes increase the components available surface area to microbes which leads to an enhanced degradation (Mohee & Mudhoo, 2005).

The output of the sorting and grinding station (C3) is sent to the primary composting vessel. Since received SSO contains leaf and yard waste, generally there is no need to provide any additional bulking agent. In both composting facilities, the over-size stream from back-end reject is used as bulking amendment if it is required. Bulking agents are used in composting to provide a proper structure for passing air and adjust the water or C/N ratio (Doublet et al., 2011; Maulini-Duran et al., 2014).

The primary composting unit implements 24 aerated containers to process material for 7-10 days in Ragged Lake facility. The primary composting in Miller facility is performed in an in-vessel composting process (Ebara technology) for 30 days. No leachate recirculation is currently implemented in the composting process (Aim Environmental Group, 2017; HRM, 2016). The constant aeration in primary composting step provides sufficient oxygen to maintain the aerobic conditions in the vessel. Another application of aeration is to remove this excess heat from the composting process (Mohee & Mudhoo, 2005).

After completion of the primary composting stage, raw compost is sent to the curing hall. Aerated floor is used in both facilities to cure the raw compost. Average curing retention time is 2.5-3 months in Ragged Lake and 3-4 weeks in Miller. The cured stream (C5) is sent to the back-end sorting unit where a screener (25 mm) is used to sort the particles size. The screener includes a vacuum blower to separate the non-composted light material and particles (e.g., light plastic films). This rejected stream is sent to landfill facility to be disposed (C10). The oversized material (>25 mm) fraction from second screening unit is recycled and used as the bulking agent. The fine fraction (C11), which is the final compost product, is sent to the transportation station to be distributed (Aim Environmental Group, 2017; HRM, 2016).

Both composting facilities are using biofilter as their air pollution control system. The gaseous output of primary composting vessels (C6). Biofilters are regularly washed and replaced, and the contaminated water (that is referred to as bio water) is collected and stored to be sent to treatment facility (Aim Environmental Group, 2017; HRM, 2016). Leachate from primary composting process along with biowater (C4 and C9 respectively) is collected and sent to wastewater treatment facility.

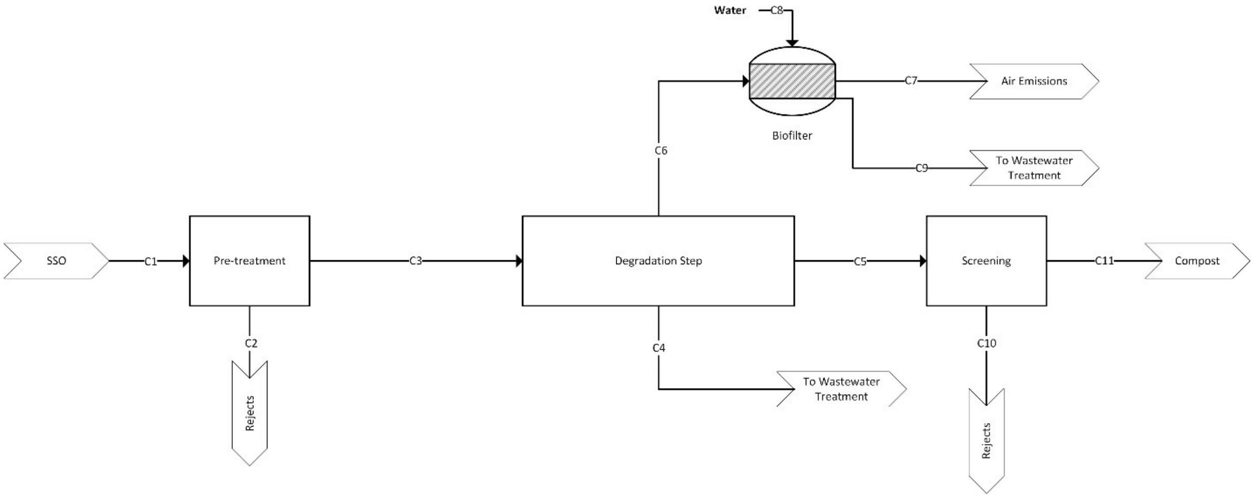

Figure 3-4 Process Flow Diagram of the composting system in HRM for both Ragged Lake and Miller central facilities. The dashed lines indicate the streams that are not included within the scope of this study

Table 3-2 Streams flowrate of Ragged Lake and Miller composting facilities

Stream	**Flowrate**		**Source**
	Ragged Lake	**Miller**	
C-1	21500	29500	Site-specific
C-2	243	333	Site-specific
C-3	21257	29167	Mass Balance
C-4	5083	6974	Site-specific
C-5	9208	12635	Mass Balance
C-6	6966	9558	Mass Balance
C-7	6966	9558	Mass Balance

C-8	114	156	(Alfonsín et al., 2013)
C-9	114	156	(Alfonsín et al., 2013)
C-10	608	835	Site-specific
C-11	8600	11800	Site-specific

Mass Flow Analysis and Emissions:

A similar mass flow analysis is used for both facilities due to similar operational condition of the facilities and lack of information on Miller facility. Site-specific data are used to the extent that was possible and supplemented by literature data whenever it was required.

The flowrate of front end and back end rejects as well as the compost product is estimated based on Ragged Lake historical data (Table 3-3). Construction and repair projects inside the facility caused fluctuation in the output of the facility, hence, an average of output to input ratios is used in this study for overall mass balance of the system. An average output ratio of 1.13 %, 2.83 % and 40 % based on weight of total received input is considered for front-end rejects, back-end rejects, and compost product respectively.

Table 3-3 Historical input and output flow rates of Ragged Lake composting facility.

Year	Total received SSO, t/y	Front-end rejects, t/y (% ww^{a} input SSO)	Back-end rejects, t/y (% ww input SSO)	Compost, t/y (% ww input SSO)	Wastewater, million liters/y	Source
2014	23,492	425 (1.81)	553 (2.35)	12,641 (53.8)	8.8	(HRM, 2016)
2015	20,250	275 (1.36)	629 (3.11)	10,296 (50.8)	8.14	(Aim Environmental Group, 2017)
2016	17,475	177 (1.01)	506 (2.90)	7,381 (42.2)	6.90	(Aim Environmental Group, 2017)
2018	20,508	180 (0.88)	596 (2.91)	6,795 (33.1)	7.57	(Aim Environmental Group, 2020)

2019	21,530	123 (0.57)	624 (2.90)	4,183 (19.4)	7.38	(Aim Environmental Group, 2019)
Avg	20,651	236 (1.13)	582 (2.83)	8259 (40)	7.76	calculated

[a] wet weight

Front-End Sorting:

Total 51,000 t of SSO is generated in Halifax in 2019. Ragged Lake facility treated approximately 21,500 t of source separated organics in 2019 (Aim Environmental Group, 2019). Since the input of the Miller facility is not reported, it is assumed that the rest of the generated SSO in HRM in 2019 is sent to Miller composting facility:

$$\text{Total generated SSO} = \text{Ragged Lake input SSO} + \text{Miller input SSO} \quad (1)$$

$$51{,}000\ \text{t/y} = 21{,}500\ \text{t/y} + Miller\ input\ SSO$$

$$\text{Miller input SSO} = 29{,}500\ \text{t/y}$$

The front-end reject flowrates for Ragged Lake and Miller are retrieved based on site-specific data from Table 3-3 and HRM report (HRM, 2016):

$$Front\ end\ Reject = Total\ input(C1) * 0.0113$$

$Ragged\ Lake$:

$$FrontEndRejects\ (C2) = 21{,}500 * 0.0113 = 243t/y$$

$Miller$:

$$Front\ End\ Rejects\ (C2) = 29{,}500\ *\ 0.0113 =\ 333t/y$$

Sorted streams flowrate is calculated based on total mass balance around the front-end sorting unit:

$$Facility\ input\ (C1) = Front\ end\ rejects\ (C2) + Sorted\ stream\ (C3)$$

$Ragged\ Lake$:

$$Sorted\ stream\ (C3) = 21{,}500\ -\ 243 = 21{,}257\ t/y$$

$Miller$:

$$Sorted\ stream\ (C3) = 29{,}500\ -\ 333 =\ 29{,}167\ t/y$$

Back-End Sorting Mass Balance:

A general mass balance is conducted on the back-end screening unit to obtain the cured compost (C10) flowrate. The back-end rejects and final compost product flowrates for

Ragged Lake and Miller are retrieved based on site-specific data from Table 3-3 and (HRM, 2016), respectively:

$Back\ end\ Reject\ (C10) = Total\ input(C1) * 0.0283$

$Ragged\ Lake$:

$Back\ end\ Reject\ (C10) = 21{,}500 * 0.0283 = 608\ t/y$

$Miller$:

$Back\ end\ Reject\ (C10) = 29{,}500 * 0.0283 = 835\ t/y$

$Final\ compost\ product\ (C11) = Total\ input(C1) * 0.40$

$Ragged\ Lake$:

$Final\ compost\ product\ (C11) = 21{,}500 * 0.40 = 8{,}600\ t/y$

$Miller$:

$Final\ compost\ product\ (C11) = 29{,}500 * 0.40 = 11{,}800\ t/y$

The back-end residues and final compost product flowrates are used in the back-end screening unit total mass balance to calculate the cured compost flowrate (C7):

$Cured\ Compost\ (C5) = Final\ product\ (C11) + Back\ end\ residues\ (C10)$

$Ragged\ Lake$:

$Cured\ Compost\ (C5) = 8{,}600 + 608 = 9208\ t/y$

$Miller$:

$Cured\ Compost\ (C5) = 11800 + 835 = 12{,}635\ t/y$

Degradation units:

The amount of leachate collected from the primary composting vessels and curing unit is adopted from the site-specific information. According to Ragged Lake 2016 annual report(NSE, 2016), total 6.90 million L wastewater were produced from processing 17,475 t SSO in 2016. Among this amount, 4.13 million L (60%) was from collected leachates while the remaining 2.77 million L (40%) was associated with the biowater (water from washing biofilters) and underground liquid piping. Considering 2016 parameters for both facilities, 0.236 m^3 leachate and 0.158 m^3 biowater is produced per t waste input. A general mass balance is conducted around degradation stage. Having the input to the composting process (C3), leachate flowrate (C4) and cured compost to screening (C5), the flowrate of the gaseous outlet (C6) is calculated:

$C3 = C4 + C5 + C6$

$Ragged\ Lake$:

$Gaseous\ Outlet$: $C6 = 21{,}253\ -\ 4{,}430\ -\ 9{,}208 = 7{,}619\ t/y$

$Miller$:

$Gaseous\ Outlet$: $C6 = 28{,}202\ -\ 2{,}520\ -\ 16{,}281 = 9{,}401\ t/y$

Air Pollution Control Unit:

The biofilter specific requirements and leachate characteristics are extracted from the available literature. The efficiency of the biofilter is not included in this study calculations as the final air emissions retrieved from the ecoinvent database are already accounted the biofiltration. No nutrient addition is considered for the biofilter as the media contains required nutrients (Colón et al., 2009; Epstein, 2011). Required water for the biofiltration is assumed 19.61 g per 1 m^3 treated air (Alfonsín et al., 2013). The amount of effluent gas that require treatment is calculated from the mass balance around the degradation unit. The mass flowrate is converted to the volumetric flowrate considering the density of process gas equal to air density (1.2 kg/m^3). Due to simplifying reasons, nutrient mass balance is not conducted on the biofilter section; as a result, the flowrate of the gaseous output is equal to the flowrate of the process gas that is sent to the biofilter. This is also the same for the liquid phase; the inlet water flowrate is equal to the output wastewater stream. The emission factors after biofiltration are estimated based on ecoinvent V3 database.

Energy Requirements:

Site specific data is used for energy consumption for each composting facility. New Era facility consumes approximately 1,129.32 MWh electricity and 35,000 liters of diesel fuel per year (52.5 kWh/t input and 1.63 l/t input). The energy consumption of Miller facility is 2,302.2 MWh electricity and 51,800 liters of diesel fuel per year (78 kWh/t input and 1.76 l/t input) (HRM, 2020b). An average of both facilities equal to 0.0653 MWh electricity per 1 t input waste and 1.8 l diesel per t waste input is considered for electricity and diesel, respectively. For diesel input, the coinvent considered 0.35 h per t waste in, with diesel consumption rate of 2.5l/ t waste in. The 0.35 h/t waste in in ecoinvent is adjusted based on site specific average of 1.8 l/t waste in, which yields to 0.25 h/t waste in.

A similar approach to EASWASTE model is used for land application background system (Hansen et al., 2006). The substitution of the commercial fertilizer is calculated separately for N, P and K. The substitution percentage is considered 30, 100 and 100% for

N, P and K, respectively. Concentration of each N, P and K are considered 1.6, 0.46, and 0.3% WW, based on experimental results on HRM compost (LP Consulting Ltd., 2017).

$$Compost\ product\ = 20400\ \frac{t\ compost}{y}$$

$$Substituted\ N\ fertilizer = 0.016 \times 20400\ \frac{t\ compost}{y} \times 0.3 = 98\ \frac{t\ N\ fertilizer}{y}$$

$$Substituted\ P\ fertilizer = 0.0046 \times 20400\ \frac{t\ compost}{y} = 94\ \frac{t\ P\ fertilizer}{y}$$

$$Substituted\ K\ fertilizer = 0.003 \times 20400\ \frac{t\ compost}{y} = 61\ \frac{t\ K\ fertilizer}{y}$$

A summary of inventory analysis for composting system is presented below:

Table 3-4 Summary of New Era composting facility inventory analysis

Stream	Unit	Ragged Lake	Miller	Total	Data type
Input					
SSO	t/y	21,500	29,500	51,000	Site Specific
Electricity	MWh/y	1,404	1,926	3,330	Site Specific
Diesel Fuel	h/y	5,375	7,375	12,750	Site Specific
Output					
Front end reject	t/y	243	333	576	Site Specific
Back-end reject	t/y	608	835	1,443	Site Specific
Total rejects	t/y	851	1,168	2,019	Site Specific
Compost Product	t/y	8,600	11,800	20,400	Site Specific
Compost Leachate	m^3/y	5083	6974	12,057	Site Specific
Biofilter Leachate	m^3/y	114	156	270	(Alfonsín et al., 2013)
Avoided N fertilizer	t/y	41.28	188.8	98	(Hansen et al., 2006; LP Consulting Ltd., 2017)

Avoided P fertilizer	t/y	39.56	54.28	94	(Hansen et al., 2006; LP Consulting Ltd., 2017)
Avoided K fertilizer	t/y	25.8	35.4	61	(Hansen et al., 2006; LP Consulting Ltd., 2017)
Air Emissions					
Carbon dioxide, non-fossil	t/y	4515	6195	10,710	ecoinvent
Ammonia	t/y	15.1	20.7	35.8	ecoinvent
Methane, non-fossil	t/y	21.7	29.8	51.5	ecoinvent
Hydrogen sulfide	t/y	1.93	2.64	4.6	ecoinvent
Dinitrogen monoxide	t/y	7.10E-01	9.74E-01	1.7	ecoinvent

3.2.2 Biosolid Processing Facility (BPF)

Process Description

Currently all dewatered sludge from WWTFs is sent to a Biosolid Processing Facility (BPF) operated by Walker Environment group, located at Aerotech industrial park. This facility is using an advanced alkaline stabilization process developed by N-Viro systems, to treat the sludge. The final product is sold as a fertilizer with Halifax Soil Amendment trademark. Different biosolids treatment processes are grouped as either Class A or Class B based on the quality of the product and its pathogen levels (Class A has higher quality than Class B). According to NS and US EPA guidelines, the N-Viro process is categorized under Class A biosolids (US EPA, 1999a, 2000).

Process flow diagram of N-viro process is shown in Figure 3-5. The N-Vrio process includes two main steps: mixing, drying and gas treatment. This process accepts different types of sewage sludge (including waste activated sludge, primary sludge, digested sludge etc.). Partially dewatered sludge, which referred to as sludge cake, is transported by truck from WWTFs and stored in the receiving area. Sludge cake (A-1) is then conveyed to a screw type mixer where Alkaline Admixture (AA) is added from an exterior silo (A-2). A wide variety of alkaline admixtures can be used in the N-Viro process including quick lime or industrial by-products such as cement kiln dust, lime kiln dust, and fly ash (N-Viro

Systems Canada Inc., 2007). Cement Kiln Dust (CMD) or Lime Kiln Dust (LKD) is used as the alkaline admixture for the Halifax N-Viro process (HRM, 2011), which are byproducts of the cement and quick lime production processes respectively. The alkaline admixture contains significant amounts of CaO, sulfate and chlorides (Arulrajah et al., 2017; Chaunsali & Peethamparan, 2013). The concentration of CaO, which is required for the alkaline stabilization process, is about 44 and 80% weight for CMD and LKD respectively (Eisa et al., 2019).

The dosage of the used alkaline admixture is one of the important process parameters in alkaline stabilization. The amount of required AA used in the HRM plant is between 30 to 40 % of sludge wet weight. This ratio is adjusted according to the amount of the heat needed for the process, type of the biosolids (especially TS content), and AA characteristics (HRM, 2011).

The output of the mixer is sent to the dryer/curing step (A-3). In dryer excess amount of moisture is removed. A mechanical rotary-drum is used as a mechanical dryer. Drying section will increase the solid concentration from 45% to 60-65% at the outlet of the drier. Dried sludge cake is sent to a curing area, called heat-pulse cell. In this cell, materials are stored for 12 hours to ensure sufficient stabilization and maximum pathogen destruction. In addition to the heat provided to the dryer, the heat generated from the chemical reaction between alkaline material and sludge maintain the temperature between 52 to 62°C. Meanwhile, the pH is controlled to slightly over 12. The elevated pH condition is kept for a total 72 hours. High temperature and pH condition is critical to remove the pathogens from the biosolids (HRM, 2011; N-Viro Systems Canada Inc., 2007). After heat-pulse cell, the cured stream can be safely stored in the storage area to be sold as the final product (A-4).

The process gas from both dryer and curing steps (A-5) is sent to the acid scrubber and then to the biofilter to remove ammonia and other potential odorous gases (HRM, 2011; N-Viro Systems Canada Inc., 2007). The generated ammonium gas is a result of the free ammonia release because of high pH conditions in the process (Bumham & Logan, 1993).

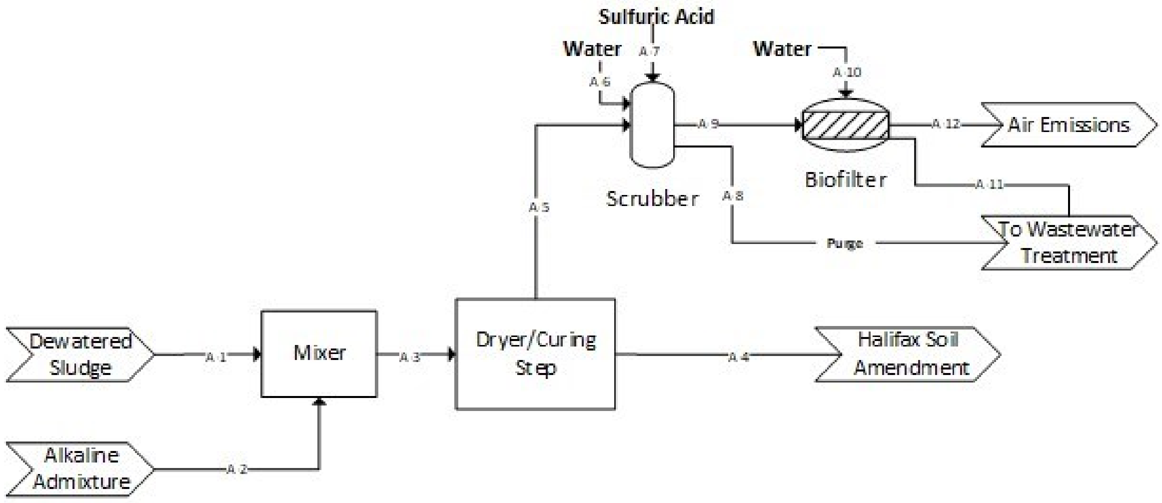

Figure 3-5 Flow diagram of alkaline stabilization facility. The bold streams are representing the streams that are used in the mass balance analysis. AA: Alkaline Admixture.

Mass Flow Analysis and Emissions

According to provided information by Halifax Water, around 30,000 t sludge is produced and sent to the BPF in 2019. The received sludge is partially dewatered and contain 28.5% TS (Halifax Water, 2020a). Assuming required Alkaline admixture (AA) dosage of 35% based on wet weight sludge (the mean value of 30-40 % range, reported by (HRM, 2011)), total 10,500 tonnes AA is consumed annually. This facility is mainly using cement kiln dust (CKD) as its alkaline admixture (HRM, 2011; Walker Industries, 2021). The CaO concentration in CKD is in the range of 38 to 60% according to literature (Eisa et al., 2019; Maslehuddin et al., 2008; Taha et al., 2004; Udoeyo & Hyee, 2002). Considering 50% CaO concentration, approximately 5250 t CaO is provided in the form of CKD, which is 17.5 % of the total sludge input or 61% of the dry matter in the input sludge. The reported amount aligns with the amount of required AA dosage reported in literature, which is in a range of 50-90 % CaO per dry matter input (Teoh & Li, 2020) or 20-30% per wet weight input sludge (Rodríguez et al., 2012; Valderrama, Granados, Cortina, et al., 2013).

Total mass balance and solids mass balance around the mixer unit is presented below:

$$\text{Total Mass Balane Around Mixer:} \qquad (2)$$

$$F_{A-1} + F_{A-2} = F_{A-3}$$

$$30{,}000 + 10{,}500 = 40{,}500 \text{ t/y}$$

$$\text{Solids Mass Balance Around Mixer:} \qquad (3)$$

$F_{A-1} \times TS_{A-1} + F_{A-2} \times TS_{A-2} \ = \ F_{A-3} \times TS_{A-3}$

$30{,}000 \times 0.285 + 10{,}500 \times 1.0 = 40{,}500 \times \ TS_{A-3}$

$TS_{A-3} = 47\ \%$

Where F_x is the flowrate of stream x (t/y), and TS_x is total solids concentration (0-1) in the stream x.

Mass balance is implemented using the final product solids concentration to calculate the gaseous effluent flowrate. Table 3-5 is showing the site-specific final output (A-4) characteristics. The TS concentration in the product is reported in range of 54.6 to 83.9 (HRM, 2011). An average number of the range, equal to 69 %, is used in the calculations for TS. This number agrees with another laboratory test performed on the facility output in 2010 (68.6% TS concentration in the product) (Hydromantis Inc., 2010) and reported numbers in the literature (minimum TS concentration of 65% for advance alkaline stabilization process) (Turovskiy & Mathai, 2006). A solid mass balance is conducted to calculate the product and process gas flowrate:

Table 3-5 Biosolid processing facility output (A-4) characteristics, retrieved from (HRM, 2011)

Parameter	Unit	Value Range	Selected Value
TS	% ww	54.6-83.9	69
VS	% TS	23-37.9	30.4
N	% TS	0.96-1.61	1.3
P (as P_2O_5)	% TS	0.94-1.73	1.3
K (as K_2O)	% TS	1.1-1.97	1.5

Solid mass balance around treatment system (4)

$F_{A-3} \times TS_{A-3} = F_{A-5} \times TS_{A-5} + \ F_{A-4} \times TS_{A-4}$

$40500 \times 0.47 = F_{A-5} \times 0 + \ F_{A-4} \times 0.69$

$F_{A-4} = 27587\ t/y$

Total mass balane around treatment system: (5)

$F_{A-3} = F_{A-5} + \ F_{A-4}$

$40{,}500 = F_{A-5} + 27587$

$F_{A-5} = 12913\ \mathrm{t/y}$

The process gas is sent to a gas treatment system with acid scrubber and biofilter. High pH condition leads to hydrogen sulfide (H_2S) reduction. This is because of conversion of H_2S to the non-volatile ionized form of HS^- (Fisher et al., 2019; Turovskiy & Mathai, 2006; WEF, 2021). As a result, no H_2S emissions is considered for the facility. Generated CO_2 as a result of decomposition of organic matters reacts with quick lime (following reaction) (Tchobanoglus et al., 2003). As a result, no biogenic CO_2 emission is considered for the process as well.

$$CO_2 + CaO \rightarrow CaCO_3 + heat$$

Ammonia (NH_3) is reported as the prominent emission in the process gas (Tchobanoglus et al., 2003). High amounts of ammonia is mainly a result of free ammonium shift to the gas form (ammonia) in response to high pH conditions (exceeding 9.3 pKa) (Fisher et al., 2019). Conversion of the free ammonia to gaseous form is shown in the following equation.

Conversion of aqueous ammonium ions:

$$CaO_{(s)} + 2NH_4^+ \leftrightarrow Ca + 2NH_3 + H_2O$$

To calculate the total amount of ammonia gas generated, it is assumed that total ammonia nitrogen in the feedstock is converted to the ammonia gas (Cartes et al., 2018). A wide range of total ammonia nitrogen is reported for sludge in the literature. Duan et al reported 783 mg/l for total ammonia nitrogen (TAN) (Duan et al., 2012a). Other studies suggested a range of 200 to 815 mg/l (van Velsen, 1979; Q. Wang et al., 2018). An average of the reported values equal to 650 mg/l TAN is considered for raw input sludge in this study. Flowrate of ammonia in the gaseous output is calculated as following:

$Free\ ammonia\ flowrate$:

$$F_{NH_3} = 30000\frac{t}{y} \times 0.000650 = 19.5\ t/y \tag{6}$$

The N-Viro system is using an acid scrubber to treat the gaseous effluent. Wet scrubbers are commonly used in organic waste processing facilities to treat process air. In a typical packed-bed tower scrubber, which is considered in this system, a countercurrent contact of odorous gas (upward) water-acid (downward) is provided to wash the air. The liquid acid-water solution is recycled while the makeup acid and water streams are provided. The purged solution is treated as wastewater and is removed periodically (Environment Canada, 2018). The ammonia scrubber is added to the system to prevent additional synthesized N_2O emissions in biofiltration step (Morris et al., 2013).

The inventory and efficiency of the acid scrubber unit is extracted from the literature. An Ammonia removal efficiency of 90% is considered for the acid scrubber unit (de Vries & Melse, 2017). For required amount of acid, 1.5 and 3.6 kg H_2SO_2 per kg removed NH_3 reported by (Costantini et al., 2020; Havukainen et al., 2022a). For this study, an average of 2.5 kg H_2SO_2 per kg removed NH_3 is assumed. A part of the scrubber effluent is recirculated to the scrubber vessel while the rest is discharged to avoid N accumulation. The amount of discharged effluent, which is referred to as purge stream (A-8), is considered 0.2 m^3 per kg NH_3 removed (Dumont, 2018).

$Amount\ of\ ammonia\ removed\ in\ wet\ scrubber$:

$$F_{\mathrm{N}H_3} = 19.5 \times 0.9 = 17.5\ t/y$$

$Amount\ of\ required\ acid$:

$$F_{\text{sulfuric acid}} = 17.5 \times 2.5 = 43.7\frac{t}{y}$$

Purge flow rate :

$$\mathrm{F}_{\text{purge,A}-8} = 17.5\frac{\mathrm{t\ N}H_3\text{ removed}}{\mathrm{y}} \times 1000\ \frac{\mathrm{kg\ N}H_3\text{ removed}}{\mathrm{t\ N}H_3\text{ removed}} \times 0.2\ \frac{\mathrm{m}^3\text{purge stream}}{\mathrm{kg\ N}H_3\text{ removed}}$$
$$= 3500\frac{\mathrm{m}^3\text{purge stream}}{\mathrm{y}}$$

As water consumption of the plant is provided by the facility, a separate water consumption for scrubber is not considered here to avoid double accounting of water requirements. Due to lack of information on the final application of the purge stream, it is assumed that it is sent to the wastewater treatment plant.

Table 3-6 Mass flow and compositional analysis for the biosolid processing facility.

Stream	Description	Flow Rate (t/y)	Resource
A-1	Dewatered sludge input	30,000	Site Specific
A-2	Alkaline admixture from the storage silo	10,500	Site Specific
A-3	Mixed alkaline admixture and sludge (sludge cake)	40,500	mass balance

A-4	Cured sludge cake, sold as Halifax soil amendment (main product)	27587	mass balance
A-5	Gaseous outlet	12,913	mass balance
A-7	Sulfuric acid to the scrubber	43.7	(Valderrama, Granados, Cortina, et al., 2013)
A-8	Acid scrubber purge, ammonia rich solution	3500	(Valderrama, Granados, Cortina, et al., 2013)
A-11	biofilter wastewater	210	mass balance

The gaseous output of the acid scrubber is sent to biofiltration for final gas treatment. The biofilter ammonia removal efficiency is assumed 90% (Colón et al., 2009):

$Ammonia\ to\ biofilteration$:

$F_{\mathrm{N}H_3,AS9} = 19.5\ - 17.5 = 2\ t/y$

$F_{\mathrm{N}H_3,emitted,A12} = 2 \times 0.1 = 0.2\ t/y = 200\ kg/y$

Plant water and energy requirements are estimated based on data provided by HRM water for 2019 (Halifax Water, 2020b). Electricity, natural gas-based heat and water are reported as 880,000 kWh/y, 2,437 GJ (62.419 m^3 NG) and 1,127 m^3 respectively.

The substitution of the commercial fertilizer is similar to composting with 30, 100 and 100% for N, P and K substitution, respectively (Hansen et al., 2006). Nutrient concentration in stabilized biosolid is considered 0.9, 0.7, and 0.6 % wet weight for NPK (Lin, 2020).

$$Soil\ Amendment\ product\ = 27587\ \frac{t}{y}$$

$$Substituted\ N\ fertilizer = 0.009 \times 27587\ \frac{t}{y} \times 0.3 = 74\ \frac{t\ N\ fertilizer}{y}$$

$$Substituted\ P\ fertilizer = 0.007 \times 27587\ \frac{t}{y} = 193\ \frac{t\ P\ fertilizer}{y}$$

$$Substituted\ K\ fertilizer = 0.006 \times 27587\ \frac{t}{y} = 165\ \frac{t\ K\ fertilizer}{y}$$

Table 3-7 Summary of Inventory Analysis for alkaline stabilization plant (biosolid processing facility).

Inventory	Unit	Value	Source
Inputs			

sludge input	tonnes/year	30,000	(Halifax Water, 2020b)
Alkaline Admixture (cement kiln dust)	tonnes/year	10,500	(HRM, 2011)
Sulfuric Acid (scrubber)	t/y	43.7	MFA
water	m^3/y	1127	(Halifax Water, 2020b)
Electricity consumption	MWh/y	880.8	(Halifax Water, 2020b)
Natural Gas consumption	GJ/y	28,161	[52]
Outputs			
Halifax Soil Amendment (product)	t/y	27,587	MFA
Scrubber purge $(NH_4)_2SO_4$ to wastewater	t/y	405	(Dumont, 2018)
Avoided N fertilizer	t/y	74.5	(Hansen et al., 2006; Lin, 2020)
Avoided P fertilizer	t/y	193.1	(Hansen et al., 2006; Lin, 2020)
Avoided K fertilizer	t/y	165.5	(Hansen et al., 2006; Lin, 2020)
Air Emissions			
NH_3	kg/y	200	MFA

3.2.3 Landfilling

Process Description

A flow diagram of the Otter Lake landfill facility process is presented in the Figure 3-6. This facility contains three main units: Front End Processing (FEP), Waste Stabilization Facility (WSF) and Residual Disposal Facility (RDF). Received waste (L-1) is sent to the FEP unit which consist of manual sorting and mechanical pretreatment units. Initially, waste undergoes a manual separation, which is conducted by staff to remove recyclables (e.g. propane tanks and other scrap metal parts) and white goods (i.e. household appliances and bulky items). The recyclables (L-2) are sent to an external recycling facility while white goods (e.g. carpet, couches, mattresses) and bulky items (L-3) are directly sent to the disposal unit.

After manual separation, sorted stream (L-4) undergoes a mechanical size adjustment, which consist of a mechanical bag breaker (MBB) and a shredder. The MBB is sorting the stream into three particle sizes: fine stream (under 50 mm), Medium stream (50-150 mm), and over-sized stream (over 150 mm). The medium-sized stream is sent to the shredder to be further shredded to under 50 mm particle sizes. Mechanical pretreatment has two main outputs: Fine particle stream (< 50mm) (L-6) and oversized stream (over 150 mm) (L-5). The fine stream, which is also referred to as the mechanically sorted organic fraction (MSOF) is transferred by conveyers to the biological stabilization unit. Oversized stream is passed to conveyers where any unacceptable material for landfill is manually removed (such as metals) and finally be disposed (SNC-Lavalin Inc., 2013).

The WSF aims to stabilize the organics before sending them to the landfill to reduce the moisture content as well as reduce the potential air emissions after disposal. The stabilization is performed by implementing a conventional aerobic semi-composting process with a series of concrete channels and turning devices. The average retention time for this aeration treatment is approximately 3 weeks. The gaseous output of the aeration vessels (L-8) is sent to biofilters before emitting to the atmosphere (L-10). The output of WSF, which is the stabilized organics (L-7), is sent to the RDF to be disposed.

RDF is a sanitary landfill equipped with low permeability base liner, leachate collection system, and landfill gas collection (with a flare system). Part of the released gas from the disposal site is captured and sent to the flaring system (L-12). No energy recovery is implemented in the facility for the flaring system and the flare output is directly released to air (L-13). Moreover, leachate is introduced to leachate collection system collects, where it is stored inleachate collection tank. The collected leachate is sent to a wastewater treatment facility (L-14) to be treated (Gartner Lee Ltd., 2007; SNC-Lavalin Inc., 2013).

Implemented combination of the mechanical and biological pre-treatment in the Otter Lake landfill (both FEP and WSF unit) before final waste disposal is referred to as the Mechanical Biological Treatment (MBT) unit (de Gioannis & Muntoni, 2007). Evidence suggests that employing a MBT unit before landfilling increases the environmental efficiency of the disposal facility through different mechanisms including reduction in gas production (in both long and short term), reduction in quantity of disposed material, reduce the leachate intensity, etc. (de Gioannis & Muntoni, 2007). The primary objective of the mechanical pre-treatment is sorting the received materials (divert the recyclables and separate the organics) and reducing the particles size. The size reduction and stabilization affects the decomposition of material in the landfill, hence decreases total emitted methane from the final disposal site (Komilis et al., 1999).

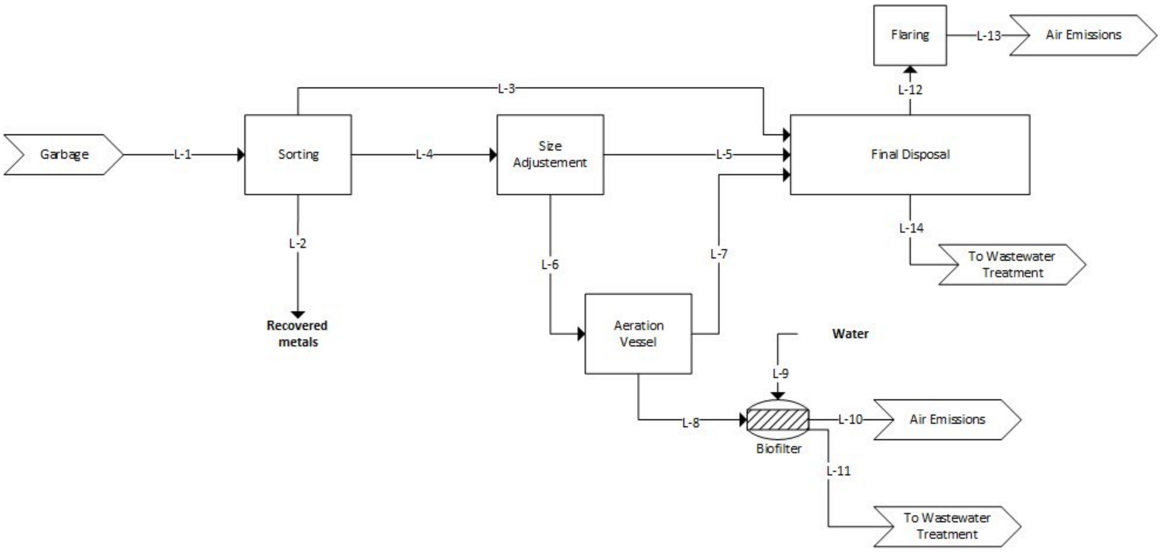

Figure 3-6 Process flow diagram of landfilling. Similar process is considered for all involved landfill facility

Mass Flow Analysis and Emissions

Initially, the efficiency of the manual separation is evaluated. Manual separation aims to separate large items and the recyclables. The amount of the diverted recyclables and large items (not compostable) is estimated based on the Otter Lake annual report, presented in the Table 3-8. 2019 values are preliminary used to estimate the process specific parameters and the flow rate analysis for the Otter Lake landfill.

Table 3-8. Otter Lake landfill general mass balance according to site specific reports in recent years. All values are in t/y and all percentage are in wet wight basis. RDF: Residual Disposal Facility, WSF: Waste Stabilization Facility, MSOF: Mechanically Sorted Organic Fraction

Year	Total garbage received, t/y	Recovered Metal, t (% garbage in)	Bulky items, L-3 (% garbage in)	Shredded to RDF, L-5 (% garbage in)	MSOF to WSF, L-6 (% ww garbage in)	Stabilized MSOF to RDF, L-7	WSF, Mass Loss, L-8 (% of ww MSOF in)	Source
2017	45,300	907 (2)	9,200 (20.3)	20,400 (44.8)	14,900 (32.9)	10,700	4,200 (28.5)	(Mirror NS, 2017)

2018	45,900	647 (1.4)	9,700 (21.1)	22,000 (48.1)	13,500 (29.4)	9,400	4,100 (30.7)	(Mirror NS, 2018)
2019	45,700	850 (1.9)	10,000 (22.0)	20,800 (45.5)	14,000 (30.6)	8,700	5,300 (37.7)	(Mirror NS, 2019)

Sorting and pre-treatment:

A range of 1.4 to 2 % of total input wet weight is recovered as metal stream with an average of 1.8 %. Metal recovery percentage is in line with the reported values in the literature, reported to be in the range of 1.7-3.6 % (Fernández-Nava et al., 2014; Montejo et al., 2013). Separation ratio of the large items and white good is also estimated based on the Table 3-8. Sorted stream flowrate from the FEP unit is calculated based on a mass balance around the manual sorting unit:

$Sorted\ waste\ stream\ flowrate\ (L-4\ stream)$:

$$F_{Sorted,L-4} = \mathrm{F_{in,L-1}} - \mathrm{F_{bulky,L-3}} - \mathrm{F_{rejects,L-2}} = 45{,}000 - 10000 - 850 = 34{,}150\ \frac{t}{y}$$

After manual sorting, the sorted mixed waste stream is sent to the mechanical bag breaker (MBB) unit. The MBB aims to adjust the size while separating the organic fraction of the mixed waste. The mechanically separated organic fraction (MSOF) is the stream with finest particles, in this unit under 50 mm. The flowrate of the MSOF stream (L-6) is determined based on the Table 3-8.

Having all outputs from FEP and mechanical treatment, the shredded mixed waste, which is referred to as residuals now on (stream L-5), is calculated by conducting a mass balance around the MBB unit:

$$F_{\mathrm{Residuals,L-5}} = F_{Sorted,\mathrm{L-4}} - F_{MSOF,\mathrm{L-6}} = 34{,}150 - 14000 = 20{,}150\ \frac{t}{y}$$

According to the presented calculations, the ratio of the separated MSOF to the residuals (L-6 to L-5) in MBB unit is 0.69. Based on the input sorted waste to mechanical treatment, 40 % of the sorted waste (L-4 stream) is separated as the MSOF (stream L-6) and the rest of it is separated as the residuals (L-5 stream). A range of 0.28 to 2.2 range is detected for the ratio of MSOF to residual streams after evaluation of more than 20 MBT plants (Połomka & Jędrczak, 2019). Basically, the flowrate of the MSOF and the residuals depends on the concentration of the organics in the mechanical treatment input waste. As a result, a highly variable organic separation ratio (ratio of the MSOF or residual stream to the mechanical treatment input) is identified.

Waste stabilization and biofilter unit:

After mechanical treatment, the separated MSOF (stream L-6) is sent to Waste Stabilization Facility (WSF) to be stabilized in aeration containers. For stabilization plant, an aerated stabilization system with short period of aeration (<4 weeks) and no further curing is considered (based on current systems in HRM). The mass reduction in the stabilization plant is a result of two mechanism: decomposition of the organic matter, and evaporation of the water content (Połomka & Jędrczak, 2019). According to site specific data (Table 3-8), a mass loss (stream L-10) in range of 28 to 38 % ww of the input MSOF with average of 32 % is identified in the Otter Lake landfill. This mass reduction is in agreement with the values presented in the literature. (Fernández-Nava et al., 2014)reported a mass reduction of 37% of the ww input MSOF at stabilization unit. (Połomka & Jędrczak, 2019) reported a total mass reduction in the range of 25 to 45 % and dry organic reduction in the range of 40 to 60 %.

To estimate C and N emissions for the WSF unit, (Boldrin et al., 2011) methodology is used. In this model, it is assumed that C degradation is relative to VS degradation, hence VS degradation percentage is used to calculate total amount of C emissions. The VS reduction after stabilization is assumed 23% after short period of aeration (lower than 4 weeks) (Scaglia et al., 2013). For Nitrogen, it is assumed that 64% of initial total Nitrogen is degraded mainly to ammonia and N_2O (de Gioannis & Muntoni, 2007). Characteristics of a typical mechanically sorted stream is used to characterize the MSOF stream (L-6). A TS concentration of 60% of wet weight, carbon concentration of 30% TS and N concentration of 0.1%Ts is considered (Cesaro et al., 2016).

$$C_{emission.} = F_{MSOF} \times TS_{MSOF} \times C \times VS_{deg}$$

$$C_{emission} = 14000 \times 0.6 \times 0.3 \times 0.23 = 580\ t/y$$

$$N_{emission.} = F_{MSOF} \times TS_{MSOF} \times N \times N_{deg}$$

$$N_{emission} = 14000 \times 0.6 \times 0.001 \times 0.64 = 5.4\ t/y$$

where C_{Emis} and N_{Emis} are total C and N air emissions, F_{MSOF} is the flow rate of the MSOF (t/y), TS_{MSOF} is the solid content of the MSOF stream (0-1) C and N are total carbon and nitrogen content of the TS (0-1), VS_{deg} is total degraded volatile solids (0-1) and N_{deg} is the total degraded nitrogen (0-1). The mass ratio of CH_4 and CO_2 is assumed 0.2 and 98.8 % of the $C_{emission}$, respectively. For nitrogen-based emissions, the ratio of major contaminants is assumed 1.5 and 98.5 % of the N_{emis}, for N_2O and NH_3 respectively(Boldrin et al., 2011).

$$CO_2\ emission = 580\frac{t}{y} \times 0.988 = 573\frac{t}{y}$$

$$CH_4\ emission = 580\frac{t}{y} \times 0.002 = 7\frac{t}{y}$$

$$N_2O\ emission = 5.4\frac{t}{y} \times 0.015 = 0.1\frac{t}{y}$$

$$NH_3 emission = 5.4\frac{t}{y} \times 0.985 = 5.3\frac{t}{y}$$

For biofilter, 90% ammonia removal per m^3 gaseous input is considered. The amount of water required for biofiltration is considered negligible compared to leachate generation.

$$NH_3 emission\ after\ biofilter\ (L-10) = 5.3\frac{t}{y} \times 0.1 = 0.53\frac{t}{y} = 530\frac{kg}{y}$$

<u>Disposal unit:</u>

Residual Disposal Facility (RDF) receives separated large items and white goods (L-3) from the front-end processing, large particles from mechanical pre-treatment (L-5) and stabilized organic fractions (L-7). Among these received streams, L-3 stream is excluded from this study as no organic fraction exist in this stream. Lon-term emissions (100-year period) is considered to model the related impacts of the final disposal. LandGem model (US EPA, n.d.) is used to estimate the amount of generated Landfill gas (LFG). LandGem uses a first-order kinetic decay equation to calculate the methane production rate each year as well as total produced LFG and total generated CO_2 (Alexander et al., 2005; Thompson et al., 2009):

$$Q_{CH_4} = \left(\frac{M}{10}\right) kL_0 e^{-kt}$$

where Q_{CH_4} is annual methane generation, k is the methane generation rate or decay rate ($year^{-1}$), L_0 is the potential methane generation capacity (m^3/t waste in), M is disposed waste flowrate (t/y), and t is the time of the waste disposal (y).

Generated methane is calculated for 2019 only to capture the impacts based on the defined functional unit, which is the impacts of the generated waste in one year. To calculate total generated LFG from both stabilized organics (L-7) and shredded mixed waste (residual stream, L-5), L_0, and k parameters is passed to LandGem.

The L_0 and k parameters require specific attention as they significantly affect the gaseous emissions results. The k parameter defines the rate of the methane generation, which depends on the decomposition rate of the degradable. A higher k values means the potential methane is emitted faster and then decay in the following years. The k value mainly depends on the moisture of the waste, availability of microorganism to degrade the organics, temperature and the pH of the waste. As a result, the k value mostly is specific to each climate region (Alexander et al., 2005). For the developed model, k is assumed 0.056 based on the Nova Scotia-specific values calculated by Thompson et al.(Thompson et al., 2009). This value is calculated based on NS average precipitation rates and decay rates.

Unlike the k value, L_0 is more waste-specific parameter (depends on the type and composition of waste) rather than region specific parameters (Alexander et al., 2005). Two waste streams, stabilized organic fraction (MSOF)(L-7) and residuals (L-5), are sent to the disposal facility with flowrates of 8700 and 21150 t/y respectively. For each of these streams, a specific L_0 value is considered from literature. The L_0 for MSOF is set 19 m^3/t waste disposed (de Gioannis et al., 2009). A ratio of 0.55 is suggested for L_0 value of residuals to L_0 value of MSOF (Buratti et al. 2015). This is basically due to much lower organic fraction in the residual as a result of mechanical pre-treatment. Considering this ratio, the L_0 is considered 10 m^3/t waste disposed for the residual stream. The results of the LandGem model for residual and MSOF streams for all facilities are presented in the Appendix A. The overall results area as follows:

$$\text{Total LFG} = \text{Residual CH}_4 + \text{MSOF CH}_4$$

$$\text{Total LFG} = \text{Captured LFG to flaring} + \text{Emitted LFG}$$

Table 3-9 Results of the LandGem model for landfill facilities

	Gaseous output	Unit	Period 1 2 y	Period 2 3 y	Period 3 35 y	Period 4 60 y	Total
Residual	Total landfill gas	t	28.72	77.12	362.18	57.32	525.33
		m^3	22999.65	61752.63	290014.40	45896.98	420663.67

	Methane	t	7.67	20.60	96.74	15.31	140.32
		m^3	11499.83	30876.31	145007.20	22948.49	210331.83
	Carbon dioxide	t	21.05	56.52	265.44	42.01	385.01
		m^3	11499.83	30876.31	145007.20	22948.49	210331.83
	NMOC[a]	t	0.05	0.13	0.62	0.10	0.90
		m^3	13.80	37.05	174.01	27.54	252.40
Stabilized	Total landfill gas	t	22.55	60.54	284.32	45.00	412.40
		m^3	18055.22	48477.14	227667.52	36030.11	330229.99
	Methane	t	6.02	16.17	75.94	12.02	110.16
		m^3	9027.61	24238.57	113833.76	18015.06	165115.00
	Carbon dioxide	t	16.53	44.37	208.37	32.98	302.24
		m^3	9027.61	24238.57	113833.76	18015.06	165115.00
	NMOC	t	0.04	0.10	0.49	0.08	0.71
		m^3	10.83	29.09	136.60	21.62	198.14

[a] NMOC: None methane oganic

Captured LFG is sent to flaring system while the emitted LFG is directly released to the atmosphere. Gas collection efficiency is considered 88% (Kirkeby et al. 2007) for all years. It is assumed that all methane in the captured LFG is burned. The emission of the flaring is calculated based on the Table 3-10. The overall gaseous emissions from disposal unit are presented in Table 3-11. Total gaseous emissions are calculated by summation of direct emissions and flared emissions.

Table 3-10 inventory data for flaring, retrieved from (Di Maria, Sordi, and Micale 2013). The units in the reference are presented based on volumetric methane rate, which is converted to mass by considering density of 0.554 kg/m^3 for methane.

Inventory	Unit	Value
NO_x	g / kg CH_4	1.14
CO	g / kg CH_4	1.33
PM	g / kg CH_4	0.43
Dioxins/furans	g / kg CH_4	1.21E-8

NMVOC removal	% removal	99.23

Table 3-11 Summary of gaseous emission from the disposal unit for 100-year period.

Emissions	Residual	Stabilized	Total
directly emitted emissions			
methane, t	16.8	13.2	30.1
CO_2,t	46.2	36.3	82.5
NMOC, kg	108.57	85.23	193.79
Flared emissions			
CO_2	338.81	265.97	604.78
NO_X, kg	141	111	251.3
CO, kg	164	129	293.2
PM, kg	53	42	94.8
NMOC, kg	5.81	4.56	10.4

In addition to gaseous emissions, long term leachate generation impact is also considered. The collected leachate is sent to wastewater treatment facility. The volume of generated leachate in landfills depends on the waste content, precipitation, amount of water run-off and evaporation ratios of the water in the soil. According to Otter Lake reports, 70400 m^3 leachate generated in 2019 (1.76 m^3 per t disposed). Assuming same amount of leachate generation each year (based on ecoinvent assumptions) for 100 years, total 7.04E6 m^3 leachate is generated.

The electricity and energy requirement are included based on site-specific data from communication with the facility. Electricity consumption of the plant is reported 180,000 kWh/month that yields 2160 MWh/y. In addition, 184,000 liters diesel per year and 86,700 liters propane per year is reported. Diesel operation hours (h) is calculated based on Ecoinvent database, considering density of 0.84 kg/l and 5.95 kg/h for diesel machine operation. Following tables show the summary of landfill facility inventory and mass balance of the facility.

Table 3-12 Mass balance table of the landfilling facilities. All values are in t/y.

Stream	Description	Flowrate	Source
L-1	Received mixed waste	45,000	Site specific
L-2	Separated metals to material recycling facility	850	Site specific
L-3	bulky items and white goods to disposal	10,000	Site specific
L-4	Manually sorted stream to mechanical bag breaker unit	34,150	MFA
L-5	Large fraction to disposal unit	20,150	MFA
L-6	Mechanically sorted Organic Fraction (MSOF): Fine particles (<50 mm), mainly organics, to aeration vessels	14,000	Site specific
L-7	Stabilized organics (MSOF) to final disposal unit	8,700	Site specific

Table 3-13 Summary of HRM landfilling inventory

Inventory	Unit	Otter Lake	Source
Inputs			
Mixed waste input	t	45,000	Site specific
Electriciy	MWh/y	2,160	Site specific
Diesel	Liters (h)	184,000 (25976)	Site specific
Propane	Liters (GJ)	86,700 (2033)	Site specific
Output			

Leachate (100 years)	m^3	7.04 E6	Site specific
Disposal Air Emissions			
Carbon dioxide, non-fossil	t	687.28	LandGem
Methane, non-fossil	t	30.1	LandGem
NMOC	Kg	204.19	LandGem
NO_x	kg	251.3	LandGem
CO	kg	293.2	LandGem
PM	kg	94.8	LandGem
WSF Air Emissions			
Methane, non-fossil	t	7	Calculations
Carbon dioxide, non-fossil	t	573	Calculations
N_2O	kg	100	Calculations
NH_3	kg	530	Calculations

3.3 Life Cycle Inventory Analysis: Novel Biorefinery

3.3.1 Process Description and Design

The novel biorefinery aims to divert the organic solid waste from conventional waste management system in HRM. A schematic of the proposed biorefinery is displayed in Figure 3-7. The proposed system is envisioned to be located in the Otter Lake landfill facility. The input of the system includes source-separated organics (SSO), mechanically sorted organic fraction of garbage (MSOF) and dewatered sludge. In the novel scenario, the SSO is diverted from composting facilities to the novel biorefinery. The MSOF stream, from landfill pre-treatment unit, is also diverted from stabilization unit to the biorefinery system. Dewatered sludge is also sent to the novel biorefinery instead of lime stabilization facility.

- SSO: 51,000 t/y
- Mechanically sorted organic fraction (MSOF) of garbage: 14,000 t/y
- Sludge: 30,000 t/y (28% TS)

Initially, all received streams are sent to feed tank (B1-3). Mechanical pretreatment is considered for SSO (B1) and sludge stream (B2) prior to sending them to the feed tank. For mechanically sorted organic fraction of garbage, the size adjustment has been already performed in the landfill facility. The size adjustment before AD enhances the solubility of the organics, therefore, increases the final methane yield (Kondusamy and Kalamdhad 2014). Currently, composting facilities in Halifax are implementing their own sorting unit to separate not-compostable material and recyclables before treatment. To reflect the waste characteristics of HRM, similar separation ratio and efficiency to composting facilities for SSO is considered in biorefinery as well. For sludge stream, no reject stream is considered. Separated non-organics (B-4) is sent to the landfill facility to be disposed.

Anaerobic co-digestion is used in the proposed novel biorefinery to treat the input streams. A mesophilic wet anaerobic co-digestion is selected for the main digestion unit. Among psychrophilic, mesophilic, and thermophilic temperature ranges (~ 25, 35 and 55 °C respectively), mesophilic range is chosen due to its higher stability, lower ammonia inhibition potential, and higher methane content in biogas (Siddique & Wahid, 2018). Anaerobic co-digestion is commonly practiced in wet and dry conditions. Wet condition is assumed in this study as it has higher methane yield and VS reduction to dry condition (A. Kumar & Samadder, 2020; Nagao et al., 2012). A typical TS concentration of 10 % of wet weight is considered for the input of the digester.

After the digestion unit, the effluent digestate (B10) is stored in a tank. Stored digestate (B11) is sent to a phase separation unit to separate solid and liquid phases. Solid-liquid separation is required for digestate valorization unit. Moreover, it decreases the transportation of solid phase cost. Phase separation also enhances the nutrient management as most of the phosphorus is accumulated in the solid phase while liquid phased contains most of the nitrogen (Guilayn et al., 2019). Different separation technologies including centrifuge, screw press and sieve belt (Duan et al., 2020). For this study, a centrifuge is chosen as it is commonly implemented in industrial anaerobic digestion units. After phase separation, the solid digestate (B12) is sent to the composting facility for further treatment. The liquid digestate (B13) is sent to the proposed digestate valorization system.

Struvite recovery is the first unit in digestate valorization system, which recover P and N as struvite via crystallization process. Magnesium source (B14) is added to the struvite recovery to shift limiting reactant from Mg to P (Hallas et al., 2019). Different Mg sources such as $MgCl_2$, $MgSO_4$, $MgCO_3$ and $Mg(OH)_2$ are suggested by literature (Hallas et al., 2019; F. Wang et al., 2019). For this research, $MgSO_4$ is chosen as the Mg source for struvite recovery. The pH is controlled in the struvite recovery process to avoid conversion of NH_4^+ to gaseous NH_3 and formed crystal size. A pH of 7.5-9 is suggested by studies for struvite crystallization of digestate (Hallas et al., 2019; F. Wang et al., 2019). Two methods have been identified to control the pH: addition of NaOH and aeration. For this study, aeration is chosen to control pH. In this method, aeration leads to degasification of the influent by stripping CO_2, which leads to pH increase.

A recycle stream (B17) is considered for the anaerobic digester. Recycling liquid digestate can increase the biogas production by increasing hydraulic residence time (HRT), leading to enhance VS degradation in digester. Recycling the liquid effluent can increase the alkalinity inside the reactor which yields to higher stability (Ni et al., 2017; Peng et al., 2016; Sayedin et al., 2019). In addition to previously indicated advantages, lower freshwater consumption (Zamanzadeh et al., 2016) and enhanced homogeneous condition inside the reactor (Méndez-Acosta et al., 2010) can be mentioned. However, excessive recycling might result in a buildup of ammonia system, which is inhibitory for the methanogens (Gottardo et al., 2017). For proposed biorefinery, two operation parameters are considered for the recycling stream: (1) recycle ratio and (2) recycling point. Recycle ratio is adjusted to avoid ammonia inhibition and maintain pH control in the reactor. Conventionally, the recirculation stream is recycled from liquid digestate to the main reactor (Méndez-Acosta et al., 2010; M.N.V Prasad, 2016; Peng et al., 2016). To lower the ammonia inhibition risk as well as avoid struvite precipitation in the main reactor, the recycling is considered after struvite recovery in the proposed biorefinery.

The ammonia recovery unit, which consists of a stripping column and an absorber column, receives the remainder of the LD after recycle stream recirculation (B18). In the stripping column, the ammonia is transferred from liquid phase to the gaseous phase by blowing a stripping agent (Walker et al., 2011). Air, nitrogen, steam, and biogas are among stripping agents used in stripping column (Serna-Maza et al., 2015). Ammonia exists in the feed in two forms: free ammonia (NH_3) and ionic ammonia (NH_4^+). Equation between ammonia forms is shown below (Campos et al., 2013):

$$NH_4^+{}_{(aq)} \leftrightarrow NH_{3(aq)} + H^+_{(aq)}$$

Two parameters can affect transferring the equation in favor of free ammonia to achieve higher ammonia removal efficiencies: temperature that affects acid ionization constant (K_a)

which is 5.39 E-10 L.mol^{-1} at 25 °C, and pH (Guštin & Marinšek-Logar, 2011). Optimum values for temperature and pH are reported above 50-70 and 9.0-10, respectively (Campos et al., 2013; Provolo et al., 2017; Serna-Maza et al., 2015; Walker et al., 2011). Another important operation parameter is the rate of the stripper agent blown to the stripping column. Stripping gas increase the accessible surface interface between gas and liquid that enhance shifting of ammonia from liquid to gas phase. Thus, higher air flowrate leads to more removal efficiencies (Walker et al., 2011).

After stripping column, the mixture of the ammonia and air is sent to an acid scrubber to recover ammonia from the gaseous form. To absorb ammonia, acid solution is sprayed or circulated through a packed bed column. Among important parameters, pH, temperature, liquid to gas ratio and the amount of ammonia in the gas can be mentioned (Havukainen et al., 2022b). In this study, sulfuric acid (B21) is considered as the absorption agent due to its common full-scale application, producing ammonium sulfate (B22) that can be used as fertilizer (Vaneeckhaute et al., 2017).

After ammonia stripping, the effluent (B23) is sent to the algae cultivation system. To prepare the effluent stream for microalgae cultivation, three factors are crucial: turbidity, carbon and nutrient (N and P) balance and controlling the cultivation to grow desired microalgae and avoid other bacteria (Tsapekos et al., 2021). An open pond raceway (OPR) system is considered for the algae cultivation due to lower complexity and cost (Chuka-ogwude et al., 2020). Microalgae specie selection highly affects produced biomass characteristics and production yield (Foteinis et al., 2018). For this study, Chlorella sorokiniana is chosen due to its successful implementation in wastewater systems (Kobayashi et al., 2013; Leite et al., 2019; Sayedin et al., 2020). The cultivated algae is then sent to the dewatering. The goal of this unit is to prepare the produced algae to be sold as the animal feed. A set of gravity thickener, centrifuge, and dryer is used to achieve final total solids. While part of the extracted water is recycled back to the digester to as treated water (B29), the rest of the separated water is sent to wastewater treatment facility (B30). The final product (B32) is sold as animal feed to replace soybean cultivation.

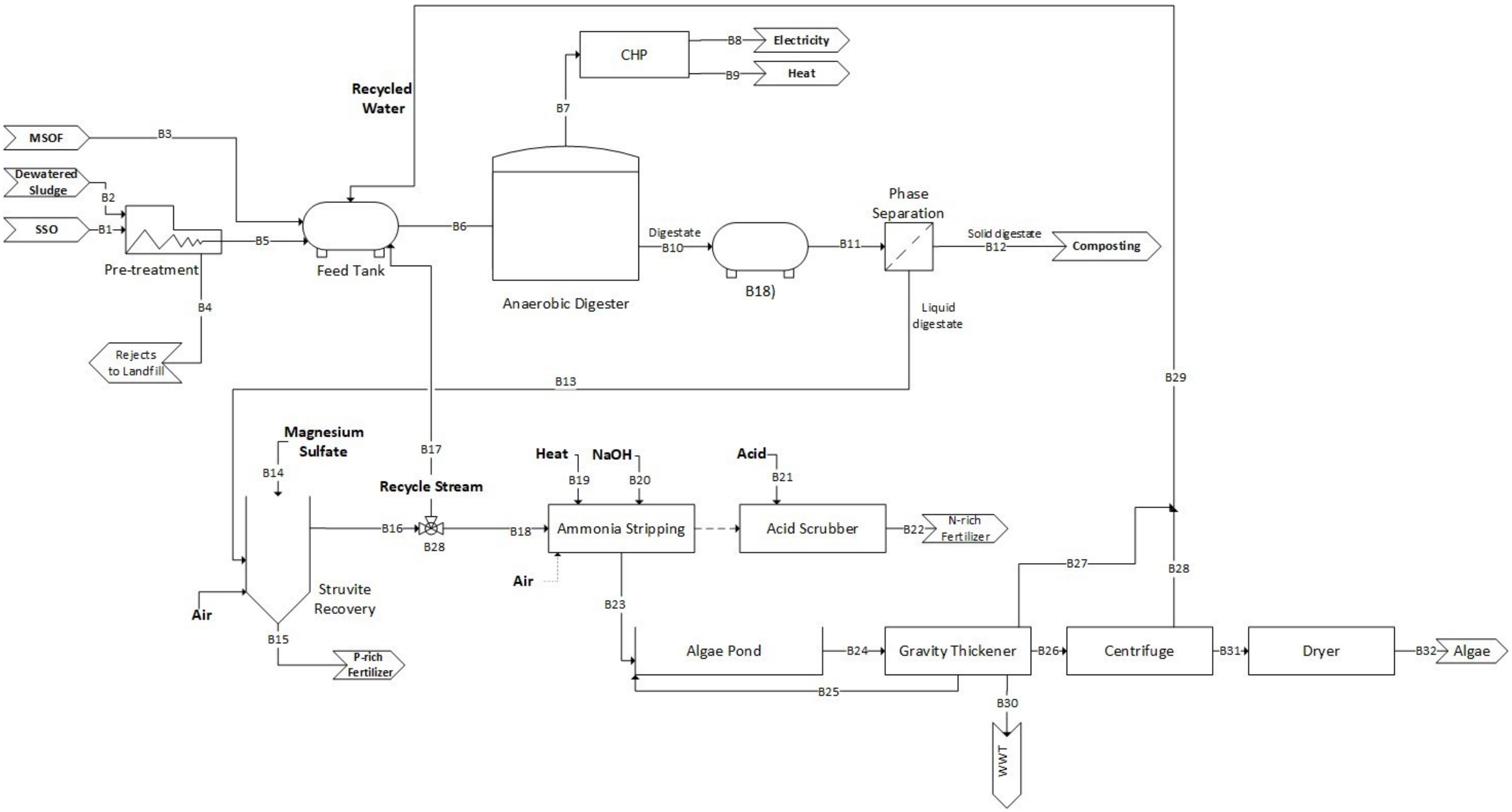

Figure 3-7 Proposed biorefinery process flow diagram.

3.3.2 Mass Flow Analysis and Emissions:

Pre-treatment and input characteristics

The input streams flowrates are considered based on site-specific information, as it was explained before. The input SSO (B1), dewatered sludge (B2), and MSOF (B3) are 51000, 30000, and 14000 t/y, respectively. Among the input streams, SSO and sludge streams are sent to the mechanical pre-treatment unit. For source-separated organics, similar removal ratio of non-degradable material in composting facilities model is considered. For Ragged Lake composting facility, 1.1% removal per t SSO input from front-end is reported. Same removal percentage is also considered for the pre-treatment of SSO in novel biorefinery that yields to separation of 561 t/y as rejects. No rejects (non-compostable material) is considered for the sludge stream.

$$\mathrm{F}_{\mathrm{rejects,B4}} = 0.011 \times F_{SSO,B1} = 0.011 \times 51000 = 561\frac{t}{y}$$

Characteristics of each input is retrieved from literature and presented **Table 3-14**, **Table 3-15**, and **Table 3-16**. Sludge composition directly depends on the wastewater treatment facility processes (L. K. Wang et al., 2008). Identifying characteristics of the sludge is more complicated than the rest input streams as the dewatered sludge in HRM is coming from different wastewater treatment processes. The received sludge stream is dewatered in wastewater treatment facilities and it contains 28% total solids (Halifax Water, 2020b). The VS concentration is also considered 66% of TS based on provided data from Mill Cove wastewater facility on their sludge for 2019 (Halifax Water, 2020a). The VS concentration aligns with the reported VS/TS according to the previous studies (Duan et al., 2012b; Edwards et al., 2017; H. Li et al., 2018).

Table 3-14 Composition of source separated organics/food waste (B1) based on literature

Parameter	Unit	Range	Selected Value	Reference
TS	%wet weight	17-37	29	(Chen et al., 2014; Davidsson et al., 2007; D.-H. Kim & Oh, 2011a; Micolucci et al., 2018; Zhang et al., 2012)

VS	%TS	81-93	86	(Chen et al., 2014; Davidsson et al., 2007; Hansen et al., 2007; D.-H. Kim & Oh, 2011a; la Cour Jansen et al., 2004; Micolucci et al., 2018; Zhang et al., 2012)
C	%TS	45-52	47	(Chen et al., 2014; Davidsson et al., 2007; Hansen et al., 2007; la Cour Jansen et al., 2004; Zhang et al., 2012)
N	%TS	2.1-3.4	2.5	(Chen et al., 2014; Davidsson et al., 2007; Hansen et al., 2007; D.-H. Kim & Oh, 2011a; la Cour Jansen et al., 2004; Zhang et al., 2012)
TAN	%wet weight	0.05	0.05	(Kim & Oh, 2011)
TP	%TS	0.3-0.6	0.4	(Davidsson et al., 2007; Hansen et al., 2007; la Cour Jansen et al., 2004; Micolucci et al., 2018; Zhang et al., 2012)
TK	%TS	0.8-1.4	1.0	(Davidsson et al., 2007; Hansen et al., 2007; Zhang et al., 2012)

Table 3-15 Characteristics of mechanically sorted organic fraction (MSOF) of garbage (B3)

Parameter	Unit	Range	Selected Value	Reference
TS	%wet weight	17.2-76	60	(Cecchi et al., 2003; Cesaro et al., 2016; de Gioannis et al., 2009)
VS	%TS	43-63	63	(Cesaro et al., 2016; de Gioannis et al., 2009; Forster-Carneiro et al., 2008)
C	%TS	23-30	30	(Cesaro et al., 2016; de Gioannis et al., 2009)
N	%TS	0.1-1.8	0.1	(Cesaro et al., 2016; de Gioannis et al., 2009)
TAN	%wet weight	0.02	0.02	(Forster-Carneiro et al., 2008)
TP	%TS	0.02-1.3	0.4	(Barampouti et al., 2019; Cecchi et al., 2003)
TK	%TS	0.56	0.56	(Huerta-Pujol et al., 2011)

Table 3-16 Characteristics of the sludge stream (B2)

Parameter	Unit	Range	Selected Value	Reference

TS	%wet weight	28	28	(Halifax Water, 2020b) (site specific)
VS	%TS	66	66	(Halifax Water, 2020a) (site specific)
C	%TS	39	39	(Edwards et al., 2017)
N	%TS	2.5-5.6	4.5	(Edwards et al., 2017; Tchobanoglus et al., 2003; L. K. Wang et al., 2008)
TAN	%wet weight	0.1	0.1	(L. K. Wang et al., 2008)
TP	%TS	0.14-2.8	1.1	(Edwards et al., 2017; Tchobanoglus et al., 2003; L. K. Wang et al., 2008)
TK	%TS	0.01-0.7	0.35	(Edwards et al., 2017; Tchobanoglus et al., 2003; L. K. Wang et al., 2008)

The output of the pre-treatment unit is calculated based on a general and parameter-specific mass balance around mechanical pre-treatment unit. It should be noted that the characteristics presented in **Table 3-14** is assumed for the stream after non-degradable (reject) extraction.

$$F_{pretreated\ input,B5} = F_{SSO,B1} + F_{biosolid,B2} - F_{rejects,B4} = 51000\frac{t}{y} + 30000\frac{t}{y} - 561\frac{t}{y} = 80439\frac{t}{y}$$

$$F_{pretreated\ input,B5} \times TS_{pretreatedinput,B5} = F_{biosolid,B2} \times TS_{biosolid,B2} + F_{treatedSSO} \times TS_{treatedSSO}$$

$$80439\frac{t}{y} \times TS_{pretreatedinput,B5} = 30000\frac{t}{y} \times 0.28 + (51000 - 561)\frac{t}{y} \times 0.29$$

$$TS_{pretreatedinput,B5} = 0.29 = 29\%\ wet\ weigh$$

Where F_x is the flowrate of the stream x in t/y, and TS_x is concentration of total solids based on wet weight (in range of 0 to 1) for stream x. The rest of the characteristics of the stream B5 is shown in **Table 3-17**. Similar approach is conducted in the rest of the sections of document to calculate the characteristics of output stream of each unit, by conducting general and parameter-specific mass balance.

Table 3-17 Characteristics of mechanical pre-treatment unit (stream B5).

Parameter	Unit	Value	Unit	Value
flowrate			t/y	80439
TS	(% ww)	29%	t/y	23027
VS	(% TS)	79%	t/y	18123
C	(% TS)	44%	t/y	10151
N	(% TS)	3.23%	t/y	744
TAN	% ww	0.07%	t/y	55

Total P	(% TS)	0.66%	t/y	151
Total K	(% TS)	0.76%	t/y	176

Anaerobic digestion Unit

A single stage wet digester is assumed for AD unit. To meet wet digestion condition, TS concentration of digester input, B6, is adjusted to 10%. The TS concentration is adjusted with processed water (B29) and recycle stream (B17). The amount of water (B29) required to reduce the TS concentration from around 40 to 10% is calculated by conducting mass balance around the feed tank, after addition of recycle stream. The recycle stream flowrate and characteristics is fixed and calculated in the next sections. The rest of the characteristics are also calculated based on mass balance:

$$
\begin{aligned}
F_{input,B6} \times TS_{input,B6} \\
&= F_{pretreated\ input,B5} \times TS_{pretreated\ input,B5} + F_{MSOF,B3} \times TS_{MSOF,B3} \\
&+ F_{processedwater,B29} \times TS_{processedwater,B29} + F_{recyycle,B17} \times TS_{recyycle,B17}
\end{aligned}
$$

$$F_{input,B6} \times 0.1 = 80439\frac{t}{y} \times 0.29 + 14000 \times 0.6 + F_{processedwater,B29} \times 0 + 175163\frac{t}{y} \times 0.0092$$

$$F_{input,B6} = 329108\frac{t}{y}$$

$$F_{processedwater,B29} = F_{input,B6} - \left(F_{pretreated\ input,B5} + F_{MSOF,B3} + F_{recyycle,B17} - F_{rejects,B4}\right)$$

$$F_{processedwater,B29} = 329108\frac{t}{y} - \left(80439\frac{t}{y} + 14000\frac{t}{y} + 174410\frac{t}{y} - 561\frac{t}{y}\right)$$

$$F_{processedwater,B29} = 60259\frac{t}{y}$$

Where F_x is the flowrate of the stream x in t/y, and TS_x is concentration of total solids based on wet weight (in range of 0 to 1) for stream x.

Table 3-18 Input characteristics to the digester after dilution and recycled stream addition (B6)

Parameter	Unit	Value	Unit	Value
flowrate			t/y	329108
TS	(% ww)	10%	t/y	32911
VS	(% TS)	74%	t/y	24363
C	(% TS)	42%	t/y	13822
N	(% TS)	4%	t/y	1345
TAN	% ww	0.17%	t/y	568
Total P	(% TS)	0.57%	t/y	187
Total K	(% TS)	2%	t/y	504

Mass flow analysis is used to calculate the output of the digestion unit based on selected process specifications. The amount of generated biogas is estimated based on Specific Gas Production (SGP) values of food waste (or organic fraction of MSW) and sludge co-digestion. The SGP value is retrieved from studies on mesophilic anaerobic co-digestion of sludge and organic waste: (Cavinato et al., 2013) evaluated the co-digestion of food waste and source separated organic waste mixed with waste activated sludge with 1:1 wet weight ratio. The SGP value of the digestion was estimated 0.35 m^3 biogas/kg VS_{fed}. In another study conducted by (Gómez et al., 2006), co-digestion of food waste and primary sludge (4:1 dry basis) was evaluated, leading to 0.5 m^3 biogas production per kg of VS_{fed}. (Zupančič et al., 2008) achieved specific biogas production of 0.6 m^3/kg VS_{fed} from co-digestion of organic fraction of MSW with primary and waste activated sludge. An average of the SPG value is used in the study, equal to 0.48 m^3 biogas/kg VS_{fed} for digester mass balance in this study.

$$\text{SGP} = 0.48\frac{m^3}{kgVS_{fed}}$$

$$VS\ concentration\ in\ feed\ =\ 74\%\ TS$$

$$TS\ concentration\ = 10\%\ wet\ weight$$

$$V_{biogas,B7}\ \text{(volumetric biogas generation)} = \text{SGP} \times F_{input.B6} \times TS_{input,B6} \times VS_{input,B6}$$

$$= 0.48\frac{m^3 biogas}{kgVS_{fed}} \times 1000\frac{kg}{t} \times 329108\frac{t}{y} \times 0.1 \times 0.74$$

$$= 1.17\ E07\frac{m^3 biogas}{y}$$

Ecoinvent V3.5 is used to calculate the inventory for the anaerobic digestion unit. The ecoinvent inventory is modified with the literature or site-specific information when it is required. A reference flow of 1 kg input waste is considered for the anaerobic digestion process in openLCA software. . Required electricity are assumed 0.00214 kWh per kg waste input based on ecoinvent database Machine operation using diesel is assumed 0.00035 h per kg waste input based on ecoinvent database for mechanical pre-treatment:

$$Electricity_{digester} = 0.00214\frac{kWh}{kg\ waste\ in} \times (51000 + 14000 + 30000 - 561)\frac{t}{y} \times 1000\frac{kg}{t}$$

$$= 2.02E05\ \frac{kWh}{y}$$

$$Machine\ operation\ (diesel) = 0.00035 \frac{h}{kg\ waste\ in} \times (51000 + 14000 + 30000 - 561) \frac{t}{y} \times 1000 \frac{kg}{t} = 3.31E04 \frac{h}{y}$$

Amount of digestate flowrate is calculated based on mass balance around digester based on calculate amount of biogas generated. The composition of biogas is retrieved from ecoinvent database (Jungbluth et al., 2007). The considered biogas consists of CH_4, CO_2, N_2, H_2S and O_2 with volumetric percentages of 67%, 32.05%, 0.70%, 0.00050%, and 0.25%. Density of biogas is considered 1.120 kg/m^3. Density of biogas, to convert volumetric flowrate to mass flowrate, is calculated based on selected composition and density of each component from ecoinvent database (Jungbluth et al., 2007):

$$\rho_{biogas,B7} = \sum \rho_x \times v_x$$

$$= \rho_{methane} \times v_{methane} + \rho_{carbon\ dioxide} \times v_{carbon\ dioxide} + \rho_{Oxygen} \times v_{oxygen} + \rho_{hydrogen\ sulfide} \times v_{hydrogen\ sulfide} + \rho_{nitrogen} \times v_{nitrogen}$$

$$= 0.714 \times 0.67 + 1.964 \times 0.32 + 1.428 \times 0.0025 + 1.517 \times 0.00001 + 1.254 \times 0.007 = 1.119 \frac{kg}{m^3}$$

Where ρ_x is density of the component x in kg per m^3, and v_x is volumetric concentration of component x (0-1).

$$\rho_{biogas,B7} = 1.119 \frac{kg}{m^3}$$

$$F_{input,B6} = F_{digestate,B10} + V_{biogas,B7} \times \rho_{biogas,B7}$$

$$329108 \frac{t}{y} = F_{digestate} + 1.17\ E07 \frac{m^3 biogas}{y} \times 1.119 \frac{kg}{m^3} \times 10^{-3} \frac{t}{kg}$$

$$F_{digestate,B10} = 316020 \frac{t}{y}$$

Where V is volumetric flow rate m^3/y, ρ is density, and F is mass flow rate in t/y.

A mass balance is conducted on the digester to determine the amount of the transferred elements and nutrients to the biogas and digestate. It is assumed that mass loss during the digestion is only due to degradation of VS. As a result, non-volatile portion of TS (ash content) remained constant after digestion. The composition of digester input was previously calculated and shown in **Table 3-18**.

$$flowrate\ of\ nonvolatile\ portion\ of\ TS: F_{nonVS} = F \times TS \times (1 - VS)$$

$$flowrate\ of\ volatile\ solids\ of\ TS: F_{VS} = F \times TS \times VS$$

$$F_{VS,input,B6} = F_{input,B6} \times TS_{input,B6} \times VS_{input,B6} = 329108 \frac{t}{y} \times 0.1 \times 0.74 = 24363 \frac{t\ VS}{y}$$

$$F_{nonVS,input,B6} = F_{input,B6} \times TS_{input,B6} \times (1 - VS_{input,B6}) = 329108 \frac{t}{y} \times 0.1 \times (1 - 0.74)$$

$$= 8548 \frac{t\ nonVS}{y}$$

VS mass balance around digester:

$$F_{VS,input,B6} = F_{VS,digestate,B10} + F_{biogas,B7}$$

$$24363 = F_{VS,digestate,B10} + 1.17\ E07 \frac{m^3 biogas}{y} \times 1.119 \frac{kg}{m^3} \times 10^{-3} \frac{t}{kg}$$

$$F_{VS,digestate,B10} = 11274 \frac{t\ VS}{y}$$

$$VS_{digestate,B10} = \frac{F_{VS,digestate,B10}}{F_{digestate,B10}} = \frac{11274}{316020} = 0.56$$

Non- volatile solid (ash) mass balance:

$$F_{nonVS,input,B6} = F_{nonVS,digestate,B10}$$

$$F_{nonVS,digestate,B10} = 8548 \frac{t\ nonVS}{y}$$

TS calculation for digestate:

$$F_{TS,digestate,B10} = F_{VS,digestate,B10} + F_{nonVS,digestate,B10} = 11274 \frac{t}{y} + 8548 \frac{t}{y} = 19823 \frac{t}{y}$$

$$TS_{digestate,B10} = \frac{F_{TS,digestate,B10}}{F_{digestate,B10}} = \frac{19823}{316020} = 0.62$$

Ammonia is produced during digestion due to degradation of organic material, especially proteins (Jiang et al., 2019). It is assumed that no ammonia transfers to biogas, as a result, no change to N concentration is considered for digestate (amount of N in input is equal to amount of N in digestate). No change in P and K concentrations is also considered after digestion as no direct reaction is considered for them. Assumption of no nutrient change after AD is in line with the previous LCA models (Y. Li et al., 2018). To account amount of ammonia generated in the digester, a typical NH_3-N/TN ratio for digestate is used based on literature. Ammonia to total nitrogen (NH_3-N/TN) is assumed 0.7 based on previous studies (Jamaludin et al., 2018; Lei et al., 2007; Schievano et al., 2011).

Total N mass balance around digester:

$$flow\ rate\ of\ N\ in\ a\ stream: F_N = F\left(\frac{t}{y}\right) \times TS\ (\%ww) \times N(\%TS)$$

$$F_{N,digestate,B10} = F_{N,input,B6}\ (no\ N\ in\ biogas\ assumption)$$

$$= 1345 \frac{t\ N}{y}$$

$$N_{digestate,B10} = \frac{F_{N,digestate,B10}}{F_{TS,digestate,B10}} = \frac{1345}{19823} \times 100 = 6.79\ \%TS$$

$$Total\ ammonia\ Nitrogen\ (TAN)\ in\ digestate\ (B10) = F_{N,digestate,B10} \times 0.7\ \frac{NH_3 - N}{TN} = 1345 \times 0.7$$

$$= 942 \frac{t\ TAN}{y}$$

Table 3-19 Characteristics of the generated digestate

Parameter	Unit	Value	Unit	Value	Unit	Value
flowrate			t/y	316020		
TS	% ww	6.27%	t/y	19823	mg/l	62726
Non-VS	%ww	2.70%	t/y	8548	mg/l	27050
VS	% TS	56.88%	t/y	11274	mg/l	35676
C	%TS	32.27%	t/y	6397	mg/l	20241
N	%TS	6.79%	t/y	1345	mg/l	4256
TAN	%ww	0.30%	t/y	942	mg/l	2979
Total P	%TS	0.94%	t/y	187	mg/l	592
Total K	%TS	0.50%	t/y	504	mg/l	1596

The digester size is estimated based on organic loading rate (OLR) and daily VS addition (Banks et al., 2011). OLR of an anaerobic co-digester depends on the ratio of the substrate that are injected. For this study, the ratio of substrates is considered equal to generated sludge and food waste in HRM region. The amount of sludge is 30000 t/y and amount of food waste (organic MSW) is 51000 SSO and 14000 MSOF. This yields to 1:2 ratio of sludge to FW, respectively. Di Maria et al (di Maria et al., 2016) reported an OLR of 2.8 kg VS/m^3 day with sludge to FW ratio of 7:2.8. Other studies suggested OLR of 5 kg VS/m^3 day for sludge to FW ratio of 7:3 (Yu et al., 2014), and 2.9 for sludge to FW ratio of 3:1 (Nghiem et al., 2017). An increase in OLR ratio is observed when the share of the higher organic concentration is added as substrate (Nghiem et al., 2017). As a result, a higher range in the studies, which is 5 kg VS/m^3 day, is considered for OLR. To convert annual input flowrate to daily flowrate, it is assumed that plant is operating continuously for 24 hours a day and 330 days a year.

$$V_{digester} = \frac{daily\ VS_{in}}{OLR}\ \frac{[kg\ VS\ day^{-1}]}{[kg\ VS\ m^{-3} day^{-1}]}$$

$$V_{digester} = 24359 \left[\frac{t\ VS}{y}\right] \times \frac{1}{330 \times 24} \left[\frac{y}{h}\right] \div 5 \left[\frac{kg\ VS}{m^3 day^1}\right]$$

$= 615m^3$

An average height/diameter ratio of 2:1 (h=2D) is considered for the digester (Moran, 2018). Achieved digester volume is align with the industrial AD plants, reported between 700 to 1200 m^3 (Bolzonella et al., 2005). A simple cylinder is considered to calculate height and diameter of the tank.

$V_{digester} = \frac{\pi D^2}{4} \times h = \frac{\pi D^3}{2} = 615m^3$

$D = 7\ m$

$h = 15\ m$

To estimate the energy requirements of the AD unit, it is assumed that the influent has the ambient temperature. The ambient temperature is assumed 8 °C based on historical monthly average temperature of Halifax, Nova Scotia (weatherspark, 2022). The specific heat capacity of input is considered 4.2 kJ/kg.°C, similar to water since the TS concentration is low (10%). Heat transfer coefficient for anaerobic digester is considered 2.5 kJ/m^2.h.K (Andreoli et al., 2017; Dhar et al., 2012; H. Li et al., 2017). The worst-case scenario is selected for the baseline, which is assuming no heat is recovered from the recycled stream and all input streams are required to heat up from ambient temperature. In addition to required heat for feed, heat loss is also considered as a contributor (Andreoli et al., 2017). Total required heat is calculated based on sum of required heat for feed and heat loss through digester.

$T_{digester} = 35\ ℃\ (mesophilic\ digester)$

$T_{feed} = 8\ ℃\ (ambient\ temperature)$

$Q_{feed} = F_{input,B6} \times C_p \times (T_{digester} - T_{feed})$

$= 329108 \times 1000 \times 4.2 \times (35 - 8)$

$= 3.75E10 \left[\frac{kJ}{y}\right] = 1.04E07 \left[\frac{kWh}{y}\right]$

Where Q_{feed} is required heat for digester in kJ/y, F_x is mass flow rate of stream x in kg/y, C_p is specific heat of the input in kJ/kg.°C, T is temperature in °C.

$A = \pi Dh = \pi \times 8 \times 17 = 336\ m^2$

$Q_{loss} = U \times A \times (T_{digester} - T_{ambient})$

$= 2.5 \times 336 \times 365 \times 24 \times (35 - 8)$

$= 1.99E8 \left[\frac{kJ}{y}\right] = 5.52E4 \left[\frac{kWh}{y}\right]$

Where Q_{loss} is the amount of heat loss from the reactor in kJ/y, U is the heat transfer coefficient in kJ/m^2.h.K, A is heat transfer area in m^2 (outer surface area, only walls are considered).

$$Q_{total} = Q_{loss} + Q_{feed} = 3.75E7\left[\frac{MJ}{y}\right]$$

Where Q_{total} is total heat required for the anaerobic digester in kJ/y. As it is shown, the heat loss has negligible contribution, and the required heat is dominantly defined by the required heat to increase the temperature of the feed to temperature of digester.

Table 3-20 proposed biorefinery summary

Parameter	Unit	Value	Source
input			
input waste (wet)	t/y	95000 (51000 t/y SSO, 14000 t/y MSOF, 30000 t/y sludge)	MFA
required electricity	kWh/y	44200	Ecoinvent
required heat	MJ/y	3.75E7	MFA
Machine operation, diesel	h	12,750	Ecoinvent
output			
biogas	m3/y	1.17E7	MFA

Combined Heat and Power (CHP) unit

Generated biogas is sent to the CHP unit to recover electricity and heat. Ecoinvent database is used to define inventory for the CHP unit (Jungbluth et al., 2007). The implemented process is “heat and power co-generation, biogas, gas engine | electricity, high voltage | Cutoff, U” where a gas engine is used to mainly generate electricity to the grid while producing heat as a co-product. Total energy efficiency is considered 0.87 for the CHP unit. This mean 87% of the total energy available in biogas is converted to either electricity or heat. Within this 87%, electrical and thermal efficiencies are considered 32 and 55% of total input energy (E_{in}), respectively. To calculate the total input energy (E_{in}), heating value of the biogas is calculated based on composition of the biogas and heating value of 35.885 and 23.413 MJ/Nm^3 for methane and hydrogen sulfide respectively (Jungbluth et al., 2007):

$$HV_{bioas} = HV_{CH_4} \times v_{CH_4} + HV_{H_2S} \times v_{H_2S} = 35.885 \times 0.67 + 23.413 \times 0.00001 = 24.043 \frac{MJ}{m^3}$$

$$E_{in} = HV_{biogas} \times V_{biogas} = 24.043 \times 1.17E07 = 2.81E08 \frac{MJ}{y}$$

$$E_{elec} = E_{total} \times Eff_{elec} = 2.81E08 \times 0.32 = 9.00E07 \ \frac{MJ_{elec}}{y}$$

$$E_{heat} = E_{total} \times Eff_{total} = 2.81E08 \times 0.55 = 1.55E08 \frac{MJ_{heat}}{y}$$

Where HV_x is the heating value of the component x and v_x is the volumetric percentage of the x component in biogas. There is no H_2S removal unit considered prior to CHP unit according to ecoinvent. The calculated numbers are shown in Table 3-21. The emissions and construction inventory are provided in Table 3-22. All ecoinvent numbers are adjusted by multiplying them by 1.17E7/0.344 so that values are representative of this study biogas rate (adjusted values are shown in the This study column).

Table 3-21 CHP unit energy inventory

Parameter	Unit	Value
Total efficiency	MJ energy recovered per MJ energy in	0.87
Energy in	MJ /y	2.81E+08
Electricity generated	MJ/y	9.00E+07
Heat generated	MJ/y	1.55E+08
normalized biogas	m^3/kWh elec generated	0.468

Table 3-22 Emission and other inputs inventory of CHP plant, retrieved from ecoinvent database. The functional unit for this study values are generated biogas per year based on HRM generated waste. The functional unit for the ecoinvent is generation of 1 kWh electricity. Each value for this study is calculated by multiplying ecoinvent values by 1.17E07 and dividing the result by 0.344. This will normalize the values based on ecoinvent database reported biogas.

Parameter	Unit	Ecoinvent	This study
Input			
biogas	m3	0.344325292	1.17E+07
heat and power co-generation unit, 160kW electrical, common components for heat+electricity	Item(s)	3.91E-08	1.33E+00

lubricating oil	kg	2.35E-04	7.98E+03
waste mineral oil	kg	-2.35E-04	-7.98E+03
Output			
Carbon dioxide, non-fossil	kg	0.65357903	2.22E+07
Carbon monoxide, non-fossil	kg	3.76E-04	1.28E+04
Dinitrogen monoxide	kg	1.96E-05	6.65E+02
Methane, non-fossil	kg	1.80E-04	6.12E+03
Nitrogen oxides	kg	1.17E-04	3.99E+03
NMVOC, non-methane volatile organic compounds, unspecified origin	kg	1.57E-05	5.32E+02
Platinum	kg	5.48E-11	1.86E-03
Sulfur dioxide	kg	1.96E-04	6.65E+03

Digestate valorization unit

Initially, digestate is sent to a centrifuge for phase separation. The solid digestate is sent to composting facility while the liquid digestate is sent to the proposed digestate valorization unit. Transfer coefficient of each component to solid and liquid rate is extracted from literature and presented in Table **3-23** for decanter centrifuge (Drosg et al., 2015; Duan et al., 2020). It is also assumed VS content of TS remains the same after separation. Required electricity for decanter is assumed 4 kWh/m^3 input (Duan et al., 2020).

Table 3-23 transfer coefficient of different parameters to liquid and solid phases in decanter centrifuge (Drosg et al., 2015; Duan et al., 2020). Conversion factors are based on mass.

Parameter	Solid phase	Liquid phase
Total flow	8%	92%
TS	86%	14%
VS	65%	35%
C	70%	30%
TN	25%	75%
TAN	7.5%	92.5%
TP	78%	22%
K	7%	93%

Characteristics of the solid and liquid digestate based on the mentioned transfer coefficients is presented in following table:

Table 3-24 Characteristics of liquid digestate after phase separation.

Parameter	Unit	Value	Unit	Value
flowrate	-		t/y	290738
TS	(% ww)	0.95%	t/y	2775
VS	(% TS)	56.88%	t/y	1578
C	(% TS)	69%	t/y	1919
N	(% TS)	36%	t/y	1009
TAN	% ww	0.30%	t/y	871
Total P	(% TS)	1.5%	t/y	41
Total K	(% TS)	16.9%	t/y	469

Table 3-25 Characteristics of Solid digestate after phase separation

Parameter	Unit	Value	Unit	Value
flowrate	-		t/y	25282
TS	(% ww)	67.43%	t/y	17047
VS	(% TS)	52	t/y	8865
C	(% TS)	26%	t/y	4478
N	(% TS)	2%	t/y	336
TAN	% ww	0.28%	t/y	71
Total P	(% TS)	0.9%	t/y	146
Total K	(% TS)	0.2%	t/y	35

Struvite recovery mass balance is calculated based on theoretical stochiometric conversion rate and literature data. Struvite crystallization reaction is presented below:

$$Mg^{2+} + NH_4^+ + PO_4^{3-} + 6H_2O \rightarrow MgNH_4PO_4.6H_2O$$

A pH of 8.5 is considered for the system based on suggested optimum pH range based on previous studies. The pH is controlled in the reactor via aeration (Styles et al., 2018). Temperature of the inlet is set to 35 °C, assuming no heat loss after digestate recovery. Total ammonia nitrogen (TAN) consists of free ammonia (NH_3) (FAN) and ammonium ion (NH_4^+). The free ammonia concentration is calculated based on the following formula (Jiang et al., 2019). It is assumed that total ammonia nitrogen (TAN) is consist of free ammonia (FAN) and ammonium ion (NH_4^+).

$pH = 8.5, T_{LD} = 36\ °C$

$$C_{FAN} = C_{TAN} \times \left(1 + \frac{10^{-pH}}{10^{-\left(0.09018 + \frac{2729.92}{T\,(K)}\right)}}\right)^{-1} = \mathrm{C_{TAN}} \times \left(1 + \frac{10^{-8.5}}{10^{-\left(0.09018 + \frac{2729.92}{(35+273)}\right)}}\right)^{-1}$$

$$= C_{TAN} \times 0.26$$

$$C_{NH_4^+} = C_{TAN} - C_{FAN} = C_{TAN} - (0.26 \times C_{TAN}) = C_{TAN} \times 0.74$$

Where C_x is mass flowrate of component x in mg/l. In order to convert the mg/l to kg/kg, density of the LD is assumed equal to water (1 kg/l).

A summary of components concentrations and flowrate are provided in Table 3-26. Due to lack of information on Mg concentration for digester input and effluent, it is assumed that all required Mg is provided from an external source. Amount of magnesium sulfate is calculated based on the kinetic equation of struvite formation. (Sayedin et al., 2019) performed experimental trials on struvite recovery from anaerobic digestate of thin stillage. In their study, the Mg:P ratio of 1:1 was maintained which is equal to theoretical molar ratio, yielded an efficiency 81% for P recovery. (Rahaman et al., 2008) investigated effects of higher Mg:P ratios (1, 1.3, and 1.6). They concluded that higher Mg:P ratio leads to better ortho-P recovery. (X. Liu et al., 2018) also showed similar results, with Mg:P ratio of 1.5:1 having highest P removal in the 8.2 pH range. Liu et al also showed 90% P removal efficiency in their experiments. A Mg:P molar ratio of 1.5:1 is considered for this study. The nutrient concentration of liquid digestate (LD) is presented below to calculate required Mg and identifying limiting agents:

$$F_{liquidDigestate,B13} = 290738\frac{t}{y} \quad (retrieved\ from\ \textbf{Table 3-24})$$

$$F_{TAN,LiquidDigestate,B13} = 871\frac{t\ TAN}{y} \quad (retrieved\ from\ \textbf{Table 3-24})$$

$$F_{NH_4^+,LiquidDigestate,B13} = 871 \times 0.74 = 644\frac{\mathrm{t\ NH_4^+}}{\mathrm{y}}$$

$$\mathrm{C}_{NH_4^+,LiquidDigestate,B13}\left(\frac{\mathrm{mg}}{\mathrm{l}}\right)$$

$$= \frac{F_{NH_4^+,LiquidDigestate,B13}\left(\frac{\mathrm{t\ NH_4^+}}{\mathrm{y}}\right)}{F_{LiquidDigestate,B13}\left(\frac{t}{y}\right)} \times 10\mathrm{E}9\frac{\mathrm{mg\ NH_4^+}}{\mathrm{t\ NH_4^+}} \times \frac{\text{1 t Liquid digestate}}{\text{1000 l liquid digestate}}$$

$$= \frac{644}{290738} \times 10^9\frac{mg}{t} \times \frac{1\ t}{1000 l} = 2216\frac{\mathrm{mg}\ NH_4^+}{\mathrm{l}}$$

$$\mathrm{M}_{NH_4^+,LiquidDigestate,B13} = \frac{\mathrm{C}_{NH_4^+,LiquidDigestate,B13}}{18.039\frac{\mathrm{mg}}{\mathrm{mmol}}} = 122.8\frac{\mathrm{mmol}\ NH_4^+}{\mathrm{l}}$$

$$F_{P,LiquidDigestate,B13} = 41\frac{\mathrm{t\,P}}{\mathrm{y}} \quad (retrieved\ from\ \textbf{Table 3-24})$$

$$\mathrm{C}_{P,LiquidDigestate,B13}\left(\frac{\mathrm{mg}}{\mathrm{l}}\right) = \frac{F_{P,LiquidDigestate,B13}\left(\frac{\mathrm{t}\,P}{\mathrm{y}}\right)}{F_{LiquidDigestate,B13}\left(\frac{t}{y}\right)} \times 10\mathrm{E}9\,\frac{\mathrm{mg}\,P}{\mathrm{t}\,P} \times \frac{\mathrm{1\,t\,Liquid\,digestate}}{\mathrm{1000\,l\,liquid\,digestate}}$$

$$= \frac{\mathrm{41\,t\,P}}{\mathrm{290738\,t}} \times 10^{9}\frac{mg}{t} \times \frac{1\,t}{1000l} = 141.5\frac{\mathrm{mg\,P}}{\mathrm{l}}$$

$$\mathrm{M}_{P,LiquidDigestate,B13} = \frac{\mathrm{C}_{P,LiquidDigestate,B13}}{30.97\frac{\mathrm{mg}}{\mathrm{mmol}}} = 4.7\frac{\mathrm{mmol}\,P}{\mathrm{l}}$$

Required Additional Mg Source:

$$\mathrm{M}_{Mg,magnesiumSource,B14} = 1.5 \times \mathrm{M}_{P,LiquidDigestate,B13} = 1.5 \times 4.7 = 7.1\frac{mmol\ Mg}{l}$$

$$\mathrm{C}_{Mg,magnesiumSource,B14} = \mathrm{M}_{Mg,magnesiumSource,B14} \times \frac{1mmol\ MgSO_4}{1\ mmol\ Mg} \times \frac{120.36\ mg\ MgSO_4}{1\ mmol\ MgSO_4}$$

$$= 7.1 \times 120.36 = 851.5\frac{mg\ MgSO_4}{l}$$

$$F_{Mg,magnesiumSource,B14} = \frac{F_{LiquidDigestate,B13} \times \mathrm{C}_{Mg,magnesiumSource,B14}}{1000000} = \frac{290738 \times 851.5}{1000000}$$

$$= 248\ \frac{t\ MgSO_4}{y}$$

Where M_x is the molar concentration of component x in mmol/l, C_x is mass concentration of component x in mg/l, F_x is flowrate of component x in t/y. Based on provided calculations, P is the limiting agent in the reaction. Regarding P removal efficiencies, (F. Wang et al., 2019) reported above 70% TP recovered when the pH is in range of 5 to 9. (Antakyali et al., 2013) evaluated large scale struvite recovery processes, indicating 85% P recovery from digested sludge in industrial scale facilities. (Hallas et al., 2019) reported a lower P recovery, in range of 41-61%, in absence of Mg addition, and above 90% in case additional Mg source is provided. Wu and Vaneeckhaute results are also aligned with other studies, suggesting a 55 to 89% recovery range for P based on different reactor configuration. For this study, a recovery rate of 90% of PO_4^{3-} (limiting reactant) is assumed for stochiometric calculations in this study with pH value of 8.5 (H. Wu & Vaneeckhaute, 2022).

Amount of recovered struvite is calculated based on stochiometric coefficient of struvite formation and P as the limiting reactant:

$$F_{struvite,B15} = 4.7\frac{mmol\ P}{l} \times \frac{1\ mmol\ struvite}{1\ mmol\ P} \times \frac{245.41\ mg\ Sturvite}{1mmol\ struvite} \times 0.90 \times \frac{10^6 l}{1\ t} \times 290738\frac{t\ liquid\ digestate}{y} = 302.8\frac{t\ struvite}{y}$$

Required electricity is 6.4E5 kJ per t struvite recovered based on industrial struvite recovery (Theregowda et al., 2019). This yields to total 1.62E+04 kWh/y electricity:

$$E_{elec} = 6.4E5\frac{kJ}{kg\ struvite} \times 302.8\frac{kg\ struvite}{y} = 5.82E07\frac{kJ}{y} = 1.62E04\frac{kWh}{y}$$

To calculate effluent N, molar N removed is considered equivalent to molar P recovered as struvite based on stochiometric coefficient:

$$F_{StruviteEffluent,B16} = F_{liquidDigestate,B13} + F_{Mg,magnesiumSource,B14} - F_{struvite,B15}$$

$$= 290738 + 248 - 302.8 = 290683\frac{t}{y}$$

Recovered N in struvite product:

$$M_{NH_4^+,struvite,B15} = 4.7\ \frac{mmolP}{l} \times 0.90 \times \frac{1\ mmolNH_4^+}{1\ mmol\ P} = 4.2\ \frac{mmolNH_4^+}{l}$$

$$C_{NH_4^+,struvite,B15} = 4.2\ \frac{mmolNH_4^+}{l} \times \frac{18\ mg\ NH_4^+}{1\ mmolNH_4^+} = 76.4\frac{mg\ NH_4^+}{l}$$

$$F_{NH_4^+,struvite,B15} = C_{NH_4^+,struvite,B15} \times\ F_{liquidDigestate,B13} = 23.9\frac{mg\ NH_4^+}{l} \times 292024\frac{t}{y} \times 1E(-6) = 22\frac{t}{y}$$

$$F_{N,StruviteRecovered,B16} = F_{N,liquidDigestate,B13} - F_{NH_4^+,struvite,B15} = 1009\frac{t}{y} - 22\frac{t}{y} = 987\frac{t}{y}$$

$$F_{TAN,StruviteRecovered,B16} = F_{TAN,liquidDigestate,B13} - F_{NH_4^+,struvite,B15} = 871 - 22 = 849\frac{t}{y}$$

Table 3-26 Input and output streams after struvite recovery

Parameter	input			Effluent		
	t/y	mg/l	mmol/l	t/y	mg/l	mmol/l
NH_4^+	653	2237	124.0	631	2161	119.8
P	41	139.9	4.7	4.09	14.0	0.5
$MgSO_4$ Added	246			-	-	-
struvite	0	0	0	300.8	1030	4.2

Table 3-27 Characteristics of struvite recovery effluent, B16

Parameter	Unit	Value	Unit	Value
flowrate	-	-	t/y	290683

TS	(% ww)	0.85%	t/y	2472
VS	(% TS)	63.84%	t/y	1578
C	(% TS)	77.62%	t/y	1919
N	(% TS)	39.91%	t/y	987
TAN	% ww	0.29%	t/y	849
Total P	(% TS)	0.17%	t/y	4.11
Total K	(% TS)	18.97%	t/y	469

After struvite recovery, part of the effluent stream is recycled back to the anaerobic digester. A wide range of recirculation ratio is reported by literature. (Gottardo et al., 2017) identified recycle ratio of 0.5-0.7, which is defined based on amount of recycled stream to amount of fresh feed (without dilution), for a pilot scale food waste anaerobic digester. Similar value is reported by (S. Wu et al., 2016), where recycle ratio is set to 0.6. Other studies defined recycle ratio of 60% (Ni et al., 2017) and 70% (Hu et al., 2014), based on fraction of the liquid digestate that is recycled. For this study, it is assumed that 60% of the struvite recovery effluent is recycled to the main digester. Concentration of COD is also compared to literature to ensure reactor is not overloaded. Since concentration of COD is not available, it is assumed that 1.4 mg COD/l is equivalent to 1 mg VS/l which is considered based on previous studies (Bullock et al., 1996; X. Liu et al., 2019). This yields to OLR of 6 kg COD/m^3.day which is inline with optimum OLR range provided by (Sayedin et al., 2019).

A summary of the input stream that is sent to digester characteristics is provided in **Table 3-18**. C/N ratio is another process parameters that is evaluated to ensure proper nutrient balance is provided for anaerobic microorganisms. According to literature, an optimum range of C/N relies in the range of 20 to 30 (M.N.V Prasad, 2016; Yen & Brune, 2007; Zahan et al., 2018). According to the input characteristics, C/N ratio is 23 for the considered HRM waste ratio, which aligns with proper C/N ratio range. The characteristics of recycling stream (B17) and stream after recycle extraction (B18) is provided in the following tables. Same concentrations to struvite recovered stream is considered for the recycling stream as no chemical reaction is performed:

Table 3-28 Characteristics of recycling stream, B17

Parameter	Unit	Value	Unit	Value

Recycle Ratio	recycled/struvite unit out	60.00%		
flowrate	-	-	t/y	174410
TS	(% ww)	0.85%	t/y	1483
VS	(% TS)	63.84%	t/y	947
C	(% TS)	77.62%	t/y	1151
N	(% TS)	39.91%	t/y	592
TAN	% ww	0.29%	t/y	509
Total P	(% TS)	0.17%	t/y	2
Total K	(% TS)	18.97%	t/y	281

After recycle stream, the remaining LD is sent to the ammonia stripping unit. According to (Guštin & Marinšek-Logar, 2011), pH plays the most critical role in ammonia removal efficiency. In addition to pH, aeration rate and temperature are the most effective parameters respectively. pH values above 10 showed less significant impact on increasing removal efficiency. As a result, pH value of 10 is considered for this study. Injected air and stream are heated before entering the stripping column to increase free ammonia (FAN) concentration. Temperature of 323 K (50 °C) is selected for the system as no significant increase in ammonia removal efficiency is reported for the higher temperatures.

Based on the mentioned pH and temperature, free ammonia is calculated. This free ammonia represents the ideal ammonia removal value (total amount of N that is ideally can be stripped). The concentration of free ammonia is calculated based on the same formula that is used in struvite recovery section (Jiang et al., 2019):

$pH = 10,\ \ T = 323\,K = 50\,°C$

$$C_{NH_3} = C_{TAN} \times \left(1 + \frac{10^{-pH}}{10^{-\left(0.09018 + \frac{2729.92}{T\,(K)}\right)}}\right)^{-1} = C_{TAN} \times \left(1 + \frac{10^{-10}}{10^{-\left(0.09018 + \frac{2729.92}{323\,K}\right)}}\right)^{-1}$$

$$\frac{C_{NH_3}}{C_{TAN}} = 0.966$$

$$F_{StrippingInput,B18} = 174410\frac{t}{y} = 174410\frac{m^3}{y}$$

$$F_{TAN,StrippingInput,B18} = 339\frac{t}{y}$$

$$C_{TAN,StrippingInput,B18} = \frac{339\frac{t\ TAN}{y}}{174410\frac{t\ solution}{y}} \times 1E6 = 2920\frac{mgTAN}{l}$$

$$C_{FAN,StrippingInput,B18} = 2920 \times 0.966 = 2822 \frac{mg\ FAN}{l}$$

$$F_{FAN,StrippingInput,B18} = 339 \times 0.966 = 328 \frac{t}{y}$$

To adjust the pH, NaOH 50% is added to the stripping column. According to (Duan et al., 2020; Styles et al., 2018), 10 kg NaOH 50% is added per m^3 liquid input. (Errico et al., 2018) reported similar amount, 7 kg NaOH 50% per 1 m^3 digestate, to bring up the pH to 9. (Vázquez-Rowe et al., 2015) reported 21.80 l NaOH per 1 tone digestate treatment. They also reported 80% liquid separation efficiency in digestate centrifuge which is used to convert the reported NaOH requirement per liquid stream enters ammonia recovery unit. For this study, an average of literature value, equal to 8.5 kg NaOH 50% per m^3 input is selected for stripper unit:

$$Required\ NaOH(B20) = 8.5\ kg \frac{NaOH}{1\ m^3 input} \times 174410 \frac{m^3 input}{y} \times \frac{t\ NaOH}{1000\ kg\ NaOH} = 988 \frac{t\ NaOH}{y}$$

For the scrubber unit, sulfuric acid 96% is added to wash the stripped air (Styles et al., 2018). (Jamaludin et al., 2018) reported that 2.37 kg acid per kg scrubbed NH_3 is required, assuming 100% ammonia recovery. (Morales et al., 2013) reported higher values, above 3.5 kg H_2SO_4 per kg N that corresponds to 4.7 kg ammonium sulfate production. A similar value is reported by (Havukainen et al., 2022b), indicating 3.6 kg H_2SO_4 per kg NH_3 removed is required. The final product is ammonium sulfate with the reported concentration of 40% w/w. For this study, a consumption rate of 3.5 kg acid per kg NH_3 is considered. A recovery efficiency of 99% is assumed based on (Zisopoulos et al., 2018). The amount of produced ammonium sulfate is calculated based on the recovery efficiency (99% input NH_3 is transferred to the air) and stoichiometric coefficients.

$$2NH_3 + H_2SO_4 \rightarrow (NH_4)_2SO_4$$

$$F_{TAN,stripped} = 0.99 \times F_{FAN,StrippingInput,B18} = 0.99 \times 328 = 324.8 \frac{t}{y}$$

$$F_{AmS,B22} = 324.8 \times \frac{1molNH_3}{17\ g} \times \frac{1molAmS}{2\ molNH_3} \times 132 \frac{gAmS}{1molAms} = 1185\ t \frac{AmS}{y}$$

The output flowrate is calculated based on a general mass balance around the ammonia stripping unit:

$$F_{StrippingEffluent,B23} = F_{FAN,StrippingInput,B18} + F_{NaOH,B20} + F_{SulduricAcid,B21} - F_{AmS,B22}$$

$$= 116273 + 988 + 1099 - 1185 = 117175 \frac{t}{y}$$

$$F_{N,StrippingEffluent,B23} = F_{N,StrippingInput,B18} - F_{TAN,stripped} = 395\frac{t}{y} - 324.8\frac{t}{y} = 70\frac{t}{y}$$

$$F_{TAN,StrippingEffluent,B23} = F_{TAN,StrippingInput,B18} - F_{TAN,stripped} = 339\frac{t}{y} - 324.8\frac{t}{y} = 14.7\frac{t}{y}$$

$$C_{TAN,StrippingEffluent,B23} = \frac{F_{TAN,StrippingEffluent,B23}}{F_{StrippingEffluent,B23}} = \frac{14.7}{117175} \times 10^9 \frac{mg}{t} \times \frac{1\,t}{1000l} = 125\ \frac{mg\ TAN}{l}$$

Table 3-29 ammonia concentration in input, effluent and product stream of the ammonia stripping unit

Stream	input		effluent	
Parameter	t/y	mg/l	t/y	mg/l
TAN	339	2920	14.7	125
FAN	328	2822	-	-
Ammonium sulfate	0	0	1185	-

The electricity required for ammonia recovery unit (stripper and scrubber) is considered 2 kWh/kg N recovered, based on (Tampio et al., 2016), which yields to 6.34 kWh/y. Electricity consumption is reported 1.1 kWh per m^3 input by (Duan et al., 2020; Styles et al., 2018) which is in similar range of the considered electricity consumption. Required heat for the ammonia stripping unit is considered 16 kWh per m^3 input, based on (Styles et al., 2018).

Table 3-30 summary of ammonia stripping unit inventory

inventory	unit	value	source
input			
electricity	kWh/y	6.50E+05	Mass balance, (Tampio et al., 2016)
NaOH 50%	t/y	988	(Duan et al., 2020; Styles et al., 2018) (Vázquez-Rowe et al., 2015)
heat	kWh/y	1.86E+06	(Duan et al., 2020; Styles et al., 2018)
Sulfuric acid	t/y	1099	Mass balance
output			
Ammonium sulfate	t/y	1185	Mass balance

Table 3-31 Characteristics of ammonia stripping unit effluent to algae pond, B23

Parameter	Unit	Value	Unit	Value
flowrate	-	-	t/y	117175
TS	(% ww)	0.84%	t/y	989
VS	(% TS)	63.84%	t/y	631
C	(% TS)		t/y	768
N	(% TS)	7.06%	t/y	70
TAN	% ww		t/y	14.7
Total P	(% TS)	0.17%	t/y	1.6
Total K	(% TS)	18.97%	t/y	188

After Ammonia stripping unit, the effluent stream is sent to the algae cultivation pound. Algae biomass concentration is calculated based on the reported biomass productivity of the C. *sorokiniana* in the literature. To ensure the media is viable for algae production, ammonia concentration is needed to be controlled as it is inhibitory for micro-algae cultivation. Ammonia concentration of 267 mg/l was determined to be the inhibitory threshold for C. *sorokiniana* by (Sayedin et al., 2020). In another study, (S. Kim et al., 2013) cultivated *sorokiniana* sp. in media with ammonia range of 10 to 160 mg/l. The calculated ammonia concentration for algae pond input in this study is 131 mg/l that aligns with the proper range reported in mentioned studies. As a result, no further treatment is considered to reduce the ammonia concentration. Another important parameter for algae cultivation is N/P ratio. (Sayedin et al., 2020) suggested that N/P ratio of 14.5 is proper for microalgae cultivation. (S. Kim et al., 2013) used media with N/P 7 to 8. (Lizzul et al., 2014) achieved biomass concentration of 220-330 mg/l (productivity of 80 to 42 mg/l/d) was achieved for C. *sorokiniana* in wastewater media with lower N/P ratio in range of 0.7-5.6. For this study, the N/P ratio is 9, which is in same range.

$$F_{TAN,StrippingEffluent,B23} = 14.7\frac{t}{y}$$

$$C_{TAN,StrippingEffluent,B23} = \frac{14.7\ \frac{t\ TAN}{y}}{117175\frac{t}{y}} \times 10^9\frac{mg}{t} \times \frac{1\ t}{1000l} = 125\frac{mg\ TAN}{l}$$

$$F_{P,StrippingEffluent,B23} = 2\frac{t}{y}$$

$$C_{P,StrippingEffluent,B23} = \frac{2\ \frac{t\ P}{y}}{117175\frac{t}{y}} \times 10^9\frac{mg}{t} \times \frac{1\ t}{1000l} = 14\frac{mg\ P}{l}$$

$$N{:}P\ ratio = \frac{130}{14} = 8.9$$

Average of algal growth is calculated based on NS location specifications using photosynthetically active radiation (PAR) and radiation use efficiency (RUE). PAR is available radiation that can be used by microalgae, usually in 400-700 nm wavelength bandwidth. To calculate PAR value, global horizontal radiation is multiplied by 0.46 that represent the fraction of radiation proper for cultivation (Clarens et al., 2010). The average hourly value of global horizontal radiation in Nova Scotia is retrieved from (CWEC, 2019; USDOE, 2022) and presented in Table 3-32. To calculate daily value of radiation, the values for each hour are summed. By multiplying daily average by number of days in each month, average monthly values are calculated.

Table 3-32 Hourly average of global horizontal radiation for Nova Scotia based on Shearwater, NS weather database.

Hourly Average	Global Horizontal Solar Radiation Wh/m^2											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	0	0	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0	0	0
6	0	0	0	7	38	69	43	11	0	0	0	0
7	0	0	7	86	121	172	138	109	40	4	0	0
8	0	15	91	201	216	261	246	227	145	83	18	0
9	48	114	207	311	332	359	359	359	261	191	99	51
10	137	216	300	386	459	477	461	475	375	283	180	143
11	223	293	379	443	524	579	540	553	466	348	225	208
12	281	366	446	472	538	594	575	600	526	365	269	239
13	290	410	491	472	529	598	614	617	547	362	275	232
14	264	367	472	443	494	595	609	547	500	320	245	190
15	207	280	422	395	445	552	563	498	425	254	185	139
16	122	207	322	339	343	481	472	432	323	175	103	66
17	30	106	216	221	270	358	368	317	206	78	15	2
18	0	10	83	118	161	233	256	204	81	5	0	0

19	0	0	0	28	66	122	117	69	3	0	0	0
20	0	0	0	0	4	24	20	1	0	0	0	0
21	0	0	0	0	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0	0	0	0	0	0	0
24	0	0	0	0	0	0	0	0	0	0	0	0
Daily Average	1602	2384	3436	3922	4540	5474	5381	5019	3898	2468	1614	1270
Monthly Average	49662	66752	106516	117660	140740	164220	166811	155589	116940	76508	48420	39370

Table 3-33 is showing summary of the total radiation, PAR, RUE and yield for each month. PAR value is calculated by multiplying monthly average radiation value by 0.46. RUE values for each month in MJ/m^2 is retrieved from (Clarens et al., 2010). Yield is calculated by multiplying PAR and RUE:

$$Yield\ \left[\frac{g}{m^2}\right] = PAR\ \left[\frac{MJ}{m^2}\right] \times RUE\ \left[\frac{g}{MJ}\right]$$

Average monthly yield is calculated by multiplying average PAR and RUE values for each month. Total annual yield is determined by summing average monthly yield. Annual yield and daily average yield are 3382 g/m^2.y and 9.23 g/m^2.d, respectively.

Table 3-33 Radiation parameters for NS

Month	Total Radiation		PAR	RUE	Monthly average yield	Daily average yield
	Wh/m^2	MJ/m^2	MJ/m^2	g/MJ	g/m^2.month	g/m^2.d
Jan	49662	178.8	82.2	0.644	53.0	1.71
Feb	66752	240.3	110.5	0.63	69.6	2.49
Mar	106516	383.5	176.4	1.13	199.3	6.43
Apr	117660	423.6	194.8	0.83	161.7	5.39
May	140740	506.7	233.1	1.26	293.7	9.47
Jun	164220	591.2	271.9	1.89	514.0	17.13
Jul	166811	600.5	276.2	1.99	549.7	17.73

Aug	155589	560.1	257.7	2.09	538.5	17.37
Sep	116940	421.0	193.7	2.4	464.8	15.49
Oct	76508	275.4	126.7	2.78	352.2	11.36
Nov	48420	174.3	80.2	1.85	148.3	4.94
Dec	39370	141.7	65.2	0.57	37.2	1.20

Total amount of available N and P for the algae cultivation are cultivation are 14.7 t/y (125 mg/l) and 2 t/y (14 mg/l), respectively. P is identified as the limiting resource for the algae cultivation. The biomass production rate is considered 2 g/l/d for this study, based on average yield from (Sayedin et al., 2020) (1.2 to 2.1 g/l/d) and (Khalid et al., 2018) (2.2 to 3.84 g/l/d). No additional C source is assumed for the algae cultivation.

$Total\ amount\ of\ algal\ biomass\ in\ the\ pond\ Effluuent$:

$$C_{Algae,PondEffluent,B24} = 2\frac{g}{l} = 2000\frac{mg}{l}$$

$$F_{Algae,PondEffluent,B24} = 2000\frac{mg\ Algae}{l} \times \frac{117175t}{y} \times \frac{tAlgae}{10^9 mg\ mg\ Algae} \times 1000\frac{l}{t} = 234.4\frac{t}{y}$$

The required land for the amount of algae production is then estimated based on the calculated dry biomass yield (9.2 g/m^2.d) and input flowrate:

$Required\ Land$:

$$F_{StrippingEffluent,B23} = 117175\frac{\mathrm{t}}{\mathrm{y}} = 117175\frac{\mathrm{m}^3}{\mathrm{y}} = 321028\frac{\mathrm{l}}{\mathrm{d}}$$

$$2\ \frac{g\ algae}{l} \times\ 321028\frac{l}{d} \times \frac{m^2.d}{9.2\ g\ algae} = 69585m^2 = 6.96\ ha$$

After cultivation, effluent stream is sent to the harvesting and dewatering unit. Since mechanical dewatering is more economical compared to thermal drying, a mechanical drying is considered before thermal drying (Molina Grima et al., 2003). The solid concentration of 89% is considered for the final product (Taelman et al., 2015). A gravity settler, a centrifuge, and a ring dryer is used for the dewatering and harvesting unit (Beal et al., 2015). For the settler, 94% algae recovery efficiency and final concentration of 20 g algae/l is considered. The effluent and concentrate flowrates are calculated based on conduction a mass balance around thickener:

$total\ algae\ from\ the\ pond$:

$$F_{Algae,PondEffluent,B24} = 234.4\frac{t}{y}$$

$recovered\ algae\ in\ concentrate\ stream$:

$$F_{Algae,SettlerConcentrate,B26} = 0.94 \times 234.4 = 220.3\frac{t}{y}$$

$algae\ concentration\ in\ concentrate\ stream$

$$C_{Algae,SettlerConcentrate,B26} = 20000\frac{mg}{l}$$

$concentrate\ flowrate$:

$$F_{SettlerConcentrate,B26} = 220.3\frac{t\ algae}{y} \times \frac{10^9\ mg\ algae}{t\ algae} \times \frac{l}{20000\ mg\ algae} \times \frac{1\ t}{1000\ l}$$

$$= 11014\frac{t}{y}$$

$effluent\ flowrate$:

$$F_{SettlerEffluent,B27,B25,B30} = total\ input - concnetrate\ flowrate = 117175 - 11014 = 106161/y$$

After the initial thickener, the concentrate is sent to the centrifuge. The algae removal efficiency of 99% and final concentration of 200 g/l for centrifuge concentrate is assumed (Beal et al., 2015). The output flowrates are calculated based on the mass balance around the centrifuge:

$$total\ algae\ from\ the\ settler = 220.3\frac{t}{y}$$

$recovered\ algae\ in\ concentrate\ stream$:

$$F_{Algae,CentrifugeConcentrate,B31} = 0.99 \times 220.3 = 218.1\frac{t}{y}$$

$algae\ concentration\ in\ concentrate\ stream$:

$$C_{Algae,CentrifugeConcentrate,B31} = 200000\frac{mg}{l}$$

$concentrate\ flowrate$:

$$F_{CentrifugeConcentrate,B31} = 218.1\frac{t\ algae}{y} \times \frac{10^9\ mg\ algae}{t\ algae} \times \frac{l}{200000\ mg\ algae} \times \frac{1\ t}{1000\ l}$$

$$= 1090\frac{t}{y}$$

$effluent\ flowrate$:

$$F_{CentrifugeEffluent,B28} = total\ input - concnetrate\ flowrate = 11014 - 1090 = 9924/y$$

The output of the centrifuge is sent to the ring dryer, to achieve 89% dry matter. The amount of evaporated water to reach desired solid concentration is calculated as following:

$input\ flowrate$: $F_{CentrifugeConcentrate,B31} = 1090\frac{t}{y}$

input algae flowrate: $F_{Algae,CentrifugeConcentrate,B31} = 218.1\frac{t}{y}$

$$output\ algae\ concentration = 0.89\ g\frac{algae}{l}$$

$$product\ flowrate = F_{AlgaeProduct,B32} = 218.1\frac{t\ algae}{y}\times\frac{10^9\ mg\ algae}{t\ algae}\times\frac{l}{890000\ mg\ algae}\times\frac{1\ t}{1000\ l}$$

$$= 245\ \mathrm{t/y}$$

$$evaporataed\ water = 1090 - 245\ = 845\ t/y$$

For this study, an average of 3.3 E-3 kWh electricity/g algae is considered for the centrifugation and race way ponds (GREET, 2011) that yields to 6.37E07 kWh/y. Required heat for the ring dryer is assumed 3556 kJ/kg evaporated water (Taelman et al., 2015) that yields to 1932455 MJ/y:

$$E_{elec} = 3.3\times10^{-3}\frac{kWh}{g\ algae\ in}\times 218.1\frac{talgae}{y}\times\frac{1E6\ g\ algae}{1\ t\ algae} = 7.27E05\frac{kWh}{y}$$

$$H_{dryer} = 3556\frac{kJ}{kg\ water\ evaporated}\times 845\frac{t\ water}{y}\times\frac{1000kg\ water}{t\ water} = 3.01E09\frac{kJ}{y}$$

Generated algae product is assumed to replace soybean cultivation that is used for animal feed. The amount of soybean is replaced by equivalent amount of protein available in algae (Beal et al., 2015). The amount of protein in algae, considering C. vulgaris as the main specie, is estimated 31% w/w, based on average range of 24-39 reported in the literature (Kobayashi et al., 2013; Sayedin et al., 2020; Singh et al., 2011). The algal protein is assumed to have similar quality of the soybean protein used for animal feed. The protein content of soybean is assumed 40% (Taelman et al., 2015). The avoided emissions from soybean cultivation is extracted from ecoinvent database Summary of co-product handling calculations for algae is presented below:

$$Protein\ from\ algae = 0.31\frac{t\ Protein}{t\ Algae}\times 218.1\frac{t\ dryalgae}{y} = 67.6\ \frac{t\ protein}{y}$$

$$Avoided\ soybean = 67.6\frac{t\ protein}{y}\times\frac{1\ t\ soybean}{0.4\ t\ protein} = 169.0\ \frac{t\ soybean}{y}$$

Mass balance of algae harvesting effluent (processed water) is provided in Table 3-34. The effluent from gravity settler and centrifuge is used as recycle water for the digester. The flowrate of the recycled processed water (B29) is assumed equivalent to the water that is needed to dilute the feed to digester (reduce TS to 10%), which was calculated in the digester section. To this mean, whole effluent from centrifuge (B28) is used as it has lower concentration of algae. The rest of the water is supplied from gravity thickener effluent. It is assumed that processed water does not contain any algae or other nutrients:

$$F_{processedwater,B29} = 60259\frac{t}{y} \quad (this\ was\ calculated\ in\ digester\ section)$$

$$F_{CentrifugeEffluent,B28} = 9924\frac{t}{y}$$

$$F_{settlerEffluent,B27} = 60259 - 9924 = 50335\frac{t}{y}$$

$$Remaining\ Settler\ effluent\ to\ WWTF$$
$$= Total\ Settler\ Effluent - Effluent\ required\ recycled\ to\ AD$$

$$F_{settlerEffluent,B30} = F_{settlerEffluent,B30+27} - F_{settlerEffluent,B27} = 106161\frac{t}{y} - 50335\frac{t}{y} = 55826\frac{t}{y}$$

Table 3-34 effluent flowrates generated in algae cultivation

Effluent stream	unit	value
gravity settler total effluent	t/y	106161
required water for AD	t/y	60259
centrifuge effluent to AD	t/y	9924
Gravity Settler to AD	t/y	50335
Remained effluent to WWT	t/y	55826

Table 3-35 Summary of algae cultivation unit inventory

input	unit	Value
electricity for pond and centrifuge	kWh/y	7.27E+05
heat for dryer	MJ/y	3.01E+06
output		
algae product	t/y	245
Avoided soybean	t/y	169
Processed water to WWTF	t/y	5.58E+04

A summary of mass balance and energy balance of the biorefinery is presented in tables Table 3-36 and Table 3-37:

Table 3-36 Summary of energy inventory for the biorefinery. Negative values for CHP unit are representing energy output (production instead of requirement)

input	unit	Value
Algae Cultivation		

Electricity (pond and centrifuge)	kWh/y	7.27E+05
Heat (dryer)	MJ/y	3.01E+06
Ammonia Stripping		
Electricity	kWh/y	6.50E+05
Heat	MJ/y	6.70E+06
Struvite Recovery		
Electricity	kWh/y	1.62E+04
Digester		
Electricity	kWh/y	2.02E+05
Heat	MJ/y	3.75E+07
CHP unit		
Generated Heat	MJ/y	-1.55E+08
Generated Electricity	kWh/y	-2.50E+07
Total		
Heat	MJ/y	-1.08E+08
Elec	kWh/y	-2.34E+07

Table 3-37 Stream flowrates in the novel biorefinery

Stream	**B1**	**B2**	**B3**	**B4**	**B5**	**B6**	**B7**	**B10**
Flowrate	51000	30000	14000	561	80439	329108	13088	316020
TS	-	8400	8400	-	23027	32911	-	19823
VS	-	5544	5292	-	18123	24363	-	11274
N	-	378	8.4	-	744	1345	-	1345
P	-	92.4	33.6	-	151	187	-	187
Stream	**B11**	**B12**	**B13**	**B14**	**B15**	**B16**	**B17**	**B18**
flowrate	316020	25282	290738	248	302.8	290683	174410	116273
TS	19823	17047	2775	-	-	2472	1483	989
VS	11274	8865	1578	-	-	1578	947	631
N	1345	346	1009	-	-	987	592	395
Total P	187	146	41	-	-	4.1	2.5	1.6
Stream	**B20**	**B21**	**B22**	**B23**	**B24**	**B26**	**B27**	**B28**
flowrate	988	1099	1185	117175	117175	11014	50335	9924

TS	-	-	-	989	234.4 (Algae)	220.3 (Algae)	-	-
VS	-	-	-	631	-	-	-	-
N	-	-	-	70	-	-	-	-
Total P	-	-	-	1.6	-	-	-	-
Stream	**B29**	**B30**	**B31**	**B32**				
flowrate	60259	55826	1090	245				
TS	-	-	218.1 (Algae)	218.1 (Algae)				
VS	-	-	-	-				
N	-	-	-	-				
Total P	-	-	-	-				

Table 3-38 Summary of the proposed biorefinery inventory

Parameter	**Unit**	**Value**	**Source**
Input			
Electricity	MJ/y	-1.08E+08	Calculation
Heat	kWh/y	-2.34E+07	Calculation
Diesel	h/y	3.31E+04	Ecoinvent
AD plant, construction	Items	0.15865	Ecoinvent
CHP plant, construction	Items	1.33	Ecoinvent
Lubricating oil (CHP)	kg	7980.0	Ecoinvent
Waste mineral oil (CHP)	kg	-7980	Ecoinvent
Magnesium sulfate	t/y	248	Calculation
sulfuric acid	t/y	1099	Calculation
NaOH 50%	t/y	988	Calculation
output			
struvite as N, avoided	t/y	302.8	Calculation
struvite as P, avoided	t/y	302.8	Calculation
ammonium sulfate, avoided	t/y	1185	Calculation
algae	t/y	245	Calculation
soybean, avoided	t/y	169.0	Calculation
effluent to wastewater facility	m^3/y	5.58E+04	Calculation

Output Emissions (from CHP)			
Carbon dioxide, non-fossil	kg/y	2.22E+07	Ecoinvent
Carbon monoxide, non-fossil	kg/y	1.28E+04	Ecoinvent
Dinitrogen monoxide	kg/y	6.65E+02	Ecoinvent
Methane, non-fossil	kg/y	6.12E+03	Ecoinvent
Nitrogen oxides	kg/y	3.99E+03	Ecoinvent
NMVOC	kg/y	5.32E+02	Ecoinvent
Platinum	kg/y	1.86E-03	Ecoinvent
Sulfur dioxide	kg/y	6.65E+03	Ecoinvent

3.3.3 Connected systems

This section investigates alteration to other waste treatment facilities in HRM in case proposed biorefinery is implemented. Landfill facility and composting facilities are considered to supplement biorefinery. The Alkaline Stabilization system, that treated the sludge, is completely removed in the novel scenario.

Landfill

The proposed biorefinery is intended to treat the mechanically sorted stream from the landfill facility. As a result, the rest of the landfill is modeled as it is discussed in the conventional system except the waste stabilization unit. Emissions from the final disposal unit is only considered for the residual waste as well. In addition, the required energy was reported from the communications with the landfill facility, as a result, it was representative of the whole landfill including the waste stabilization system. The required electricity of the aeration system is retrieved from the literature and deducted from the total reported electricity requirement of the facility. According to (Fernández-Nava et al., 2014), 65.9 kWh per t waste input is required for the stabilization system. Considering 14000 t/y input MSOF, estimated electricity requirement for the stabilization unit is 922600 kWh/y (923 MWh/y). Total required electricity for the facility is reported 2160 MWh/y. Deducting the stabilization unit from it results in energy consumption of 1237 MWh/y. Any potential diesel consumption for the stabilization unit is neglected.

Table 3-39 Emissions from disposal unit of the landfill for the novel biorefinery scenario

Emissions	Value
directly emitted emissions	

methane, t	16.8
CO_2, t	46.2
NMOC, kg	108.57
Flared emissions	
CO_2	338.81
NO_X kg	141
CO kg	164
PM, kg	53
NMOC, kg	5.81

An average of 1.76 m^3 leachate was assumed to be generated per tonne of waste disposed based on the average plant leachate generation (discussed in the baseline scenario). In the conventional scenario, total 38,850 t/y treated waste is disposed. Among this disposed waste, 8,700 t/y is stabilized waste which is diverted from landfill in the novel biorefinery scenario. This yields to 30,150 t/y total disposed waste in the novel scenario. With the estimated leachate generation of 1.76 m^3/t waste disposed, approximately, 53000 m^3 leachate is generated each year, which is equal to 5.3E+6 m^3 leachate in 100-year time span. Summary of total landfill inventory for the novel biorefinery scenario is presented below:

Table 3-40 Summary of inventory for landfilling facility in NB scenario

Inventory	Unit	Otter Lake	Source
Inputs			
Mixed waste input	t	45,000	Site specific
Electricity	MWh/y	1237	Site specific
Diesel	Liters (h)	184,000 (25976)	Site specific
Propane	Liters (GJ)	86,700 (2033)	Site specific
Output			
Leachate (100 years)	m^3	5.3 E6	Site specific
Disposal Air Emissions			
CO_2	t	385.0	LandGem
Methane	t	16.8	LandGem
NMOC	Kg	114.4	LandGem

NO_x	kg	141	LandGem
CO	kg	164	LandGem
PM	kg	53	LandGem

Solid digestate composting

Recovered solid digestate (SD) from the phase separation is sent to the composting facility. The characteristics of the solid digestate stream is represented in Table 3-25. A compost production rate of 0.4 t compost/t solid digestate is considered based on ecoinvent database:

$$Compost\ product\ = 0.4 \times 25282 \frac{t\ SD}{y} = 10113 \frac{t\ compost}{y}$$

As it was presented in Table 3-25, solid digestate nutrients concertation are 2 %TS (1.3% wet weight), 0.9 %TS (0.6 %wet weight), and 0.2% TS (0.13 %wet weight) for N, P and K, respectively. A total nitrogen loss of 34% is considered during the composting based on (Manu et al., 2021; van Haaren et al., 2010). This yields to 0.9 % wet weight nitrogen concentration in the final compost product. No nutrient loss is considered for P and k. Similar to composting section, fertilizer substitution is considered 30%, 100%, and 100% for N, P and K respectively.

$$Substituted\ N\ fertilizer\ by\ compost = 0.009 \times 10113\ \frac{t\ compost}{y} \times 0.3 = 27\ \frac{t\ N\ fertilizer}{y}$$

$$Substituted\ P\ fertilizer\ by\ compost = 0.006 \times 10113\ \frac{t\ compost}{y} = 61\ \frac{t\ P\ fertilizer}{y}$$

$$Substituted\ K\ fertilizer\ by\ compost = 0.0013 \times 10113\ \frac{t\ compost}{y} = 13\ \frac{t\ K\ fertilizer}{y}$$

The rest of the composting parameters, including energy consumption and leachate generation is assumed similar to the composting model that was discussed in the HRM scenario. It is also assumed that most of the diesel consumption is for shredding and sorting, which is not required for the SD composting. As a result, the diesel inventory is removed from solid digestate composting model. A summary of the inventory used for solid digestate composting is presented in following table:

Table 3-41 Summary of composting unit for solid digestate

Stream	Unit	Value	Data type

Input			
Solid digestate	t/y	25282	Site Specific
Electricity	MWh/y	1.67E+03	Site Specific
Output			
Compost Product	t/y	10113	Site Specific
Compost Leachate	m^3/y	6.17E+03	Site Specific
Avoided N fertilizer	t/y	27	MFA
Avoided P fertilizer	t/y	61	MFA
Avoided K fertilizer	t/y	13	MFA
Air Emissions			
Carbon dioxide, non-fossil	t/y	5.56E+03	ecoinvent
Ammonia	t/y	1.77E+01	ecoinvent
Methane, non-fossil	t/y	2.53E+01	ecoinvent
Hydrogen sulfide	t/y	1.33E+01	ecoinvent
Dinitrogen monoxide	t/y	6.32E-01	ecoinvent

3.4 Life Cycle Inventory Analysis: Background Systems

3.4.1 Waste Collection and Transportation

The emission from waste collections and product distribution transportation is considered in this section. Separate trucks are used in HRM to collect SSO and garbage from residentials and ICI. Collected SSO is directly transported to composting facilities, Miller and New Era, while the garbage is sent to Otter Lake landfill facility.

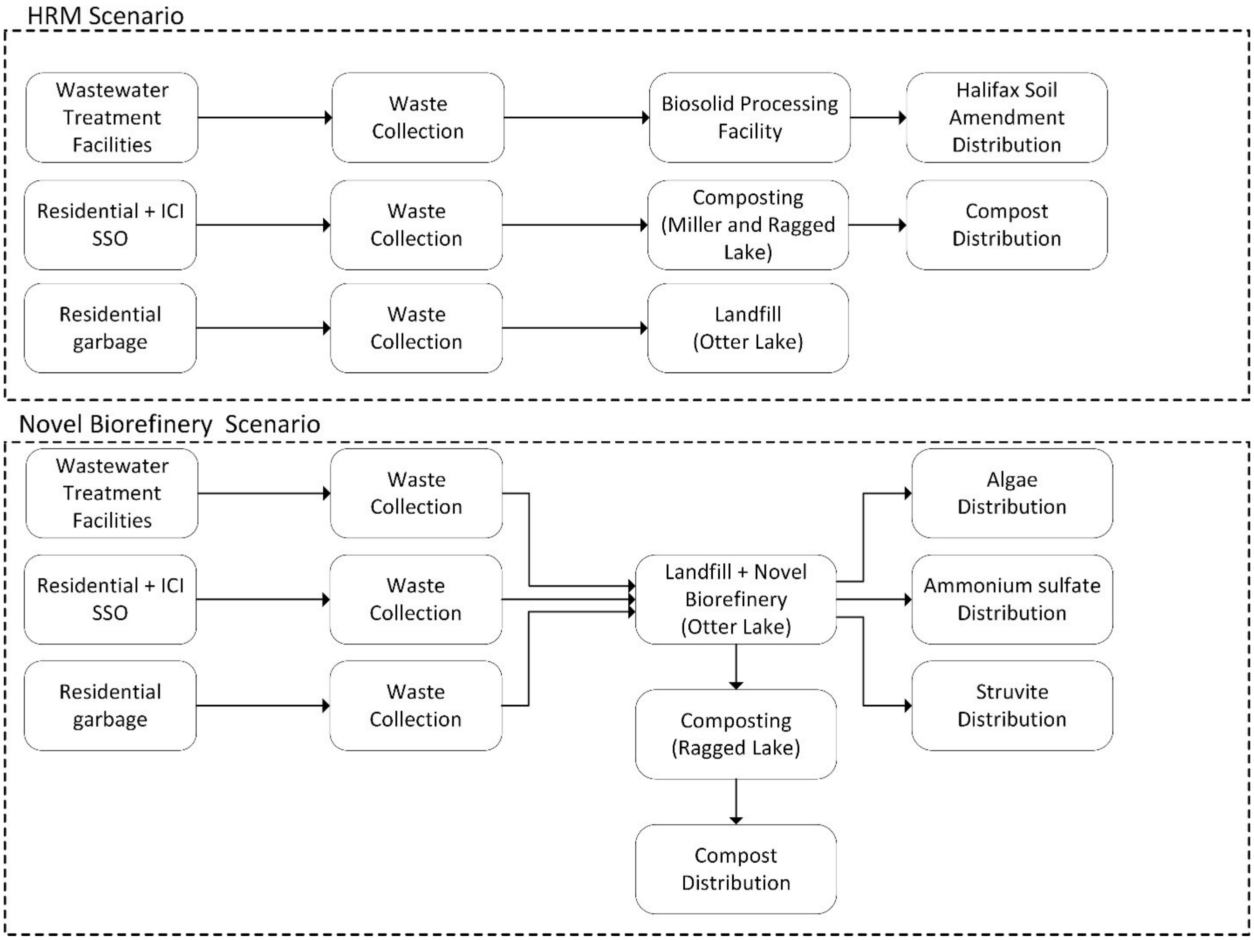

Figure 3-8 Schematic of transportation required for HRM and proposed scenario. Considered transportation include waste collection and generated product distribution.

To estimate the waste collection and transportation emissions for SSO and garbage streams, census 2016 tracts population database is used (HRM 2017) (**Figure 3-9**). In this database, HRM is divided into 92 main districts with population available for each district. To calculate the rough estimate of collection distance, the route from center of each district polygon to treatment facility is calculated (**Figure 3-10**). The amount of collected waste from each district is estimated by normalizing total generated waste in HRM based on population of each district. The details of each district calculation results are presented in Appendix B. For sludge, the transportation route from each of 14 wastewater treatment facilities (WWTF) to the biosolids processing facility is calculated. The amount sludge generated in each facility is normalized by the amount of water treated in one year, retrieved from (Halifax Water 2017). The details of sludge transportation routes are also provided in Appendix B.

$$Freight\ Transportation\ (t.km) = \sum_{i=1}^{n} Route_i \times Waste_i$$

Where freight transportation is the inventory that is used in LCA to estimate the transportation emissions (t*km), n is number of routes for each transportation phase (n=92 for garbage and SSO collection and n=14 for sludge transportation from WWTFs), and $waste_i$ is amount of waste that is collected from the source i.

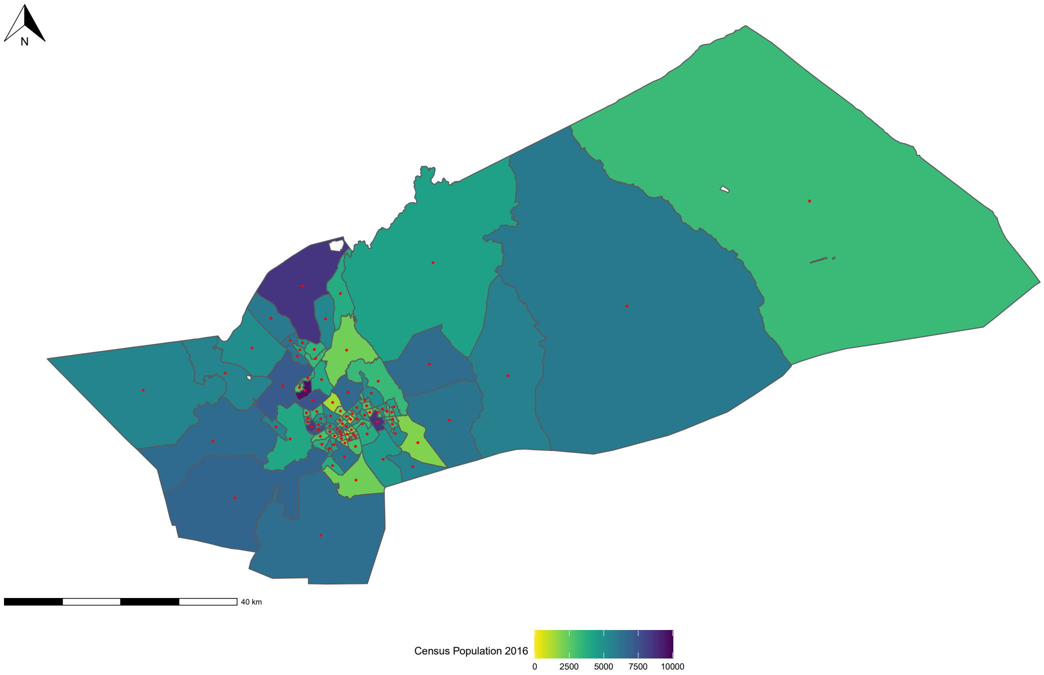

Figure 3-9 HRM census 2016 population distribution map in 92 districts. Red dots are indicating the center of each district polygon that is used to estimate the destination route of each district to the waste treatment facility.

A summary of the waste collection distances, and inventory is presented in Table 3-42. For both HRM scenario, waste collection emission inventory includes collection of source separated organics (SSO) and garbage from 92 districts to the treatment facilities. Collected garbage from residentials, equal to 45000 t/y, is sent to Otter Lake landfill. For source separated organics (SSO), the collected waste stream is sent to either Miller facility (Dartmouth) or New Era facility (Halifax). Since no information is available on which of these two facilities are serving which district, it is assumed that collected SSO is sent to the closest facility to each district. The transportation inventory also includes distribution of products, which are compost (from composting facilities) and Halifax soil amendment (from biosolid processing facility). For product distributions, it is assumed all produced products are sent to local farms. The distance of the farms is estimated in range of 75 to 100 km from the composting facilities in HRM (LP Consulting Ltd., 2017). The higher range, 100 km, is considered the distribution distance for all facilities. Details of these information can be found in Appendix B.

$$Compost\ distribution = compost\ flowrate \times distance = 20400 \frac{t}{y} \times 100\ km = 2040000\ t * km$$

$$Soil\ amendment\ distribution = Soil\ amendment\ flowrate \times distance = 27587 \frac{t}{y} \times 100\ km$$
$$= 2758700\ t * km$$

For the novel biorefinery scenario, similar garbage collection to the HRM scenario is considered while SSO collection inventory is different as the SSO is diverted from composting facilities to the biorefinery (detailed information is provided in Appendix B). The transportation of solid digestate form novel biorefinery to the composting facility is also included in the NB scenario. Between Miller and Ragged Lake composting facilities, Ragged Lake is chosen as it is closer to the otter lake landfill (location of novel biorefinery). Multiplying distance (9 km) to 25,522 t/y solid digestate, yields 229,698 t*km as the transportation inventory. The distribution distance to farms (100 km) for the other by-products, such as struvite, ammonium sulphate (AmS), and algae, is also taken into account (similar to HRM scenario):

$$Solid\ digestate\ inventory = solid\ digestate\ flowrate \times distance = 25282 \frac{t}{y} \times 9\ km$$
$$= 227534\ t * km$$

$$Compost\ (NB\ Scenario) inventory = compost\ flowrate \times distance = 10113 \frac{t}{y} \times 100\ km$$
$$= 1011300\ t * km$$

$$Struvite\ inventory = Struvite\ flowrate \times distance = 302.8 \frac{t}{y} \times 100\ km = 30280\ t * km$$

$$AmS\ inventory = AmS\ flowrate \times distance = 1185 \frac{t}{y} \times 100\ km = 118500\ t * km$$

$$Algae\ inventory = Algae\ flowrate \times distance = 210.3 \frac{t}{y} \times 100\ km = 21030\ t * km$$

To calculate the transportation emissions, ecoinvent database is used. For garbage and SSO collection, “municipal waste collection service by 21 metric ton lorry” process is used while for the rest of the transportation, “transport, freight, lorry, unspecified” is used.

. Details of calculation is presented in Appendix B.

Table 3-42 Summary of transportation inventory for HRM and novel biorefinery (NB) scenarios.

Waste Stream	**Source**	**Destination**	**Inventory (t*km)**
HRM Scenario			
Garbage collection	generation source	Otter lake landfill	1.16E+06
SSO collection	generation source	Miller and Ragged Lake composting	9.04E+05
sludge collection	WWTF	Biosolid processing facility	1.10E+06
Compost distribution	Miller and Ragged Lake	NS farms	2.04E+06
Soil amendment distribution	Biosolid processing facility	NS farms	2.76E+06
Total	-	-	7.95E+06
Novel Biorefinery Scenario			
Garbage collection	generation source	Otter lake landfill	1.16E+06
SSO transportation	generation source	Otter lake landfill (novel biorefinery)	1.31E+06
sludge collection	WWTF	Otter lake landfill (novel biorefinery)	5.75E+05
Solid digestate transportation	Otter lake landfill (novel biorefinery)	Ragged Lake Composting	2.28E+05
Struvite distribution	Otter lake landfill (novel biorefinery)	NS farms	3.03E+04
AmS distribution	Otter lake landfill (novel biorefinery)	NS farms	1.19E+05
Algae distribution	Otter lake landfill (novel biorefinery)	NS farms	2.10E+04
Compost distribution	Ragged Lake	NS farms	1.01E+06
Total			4.45E+06

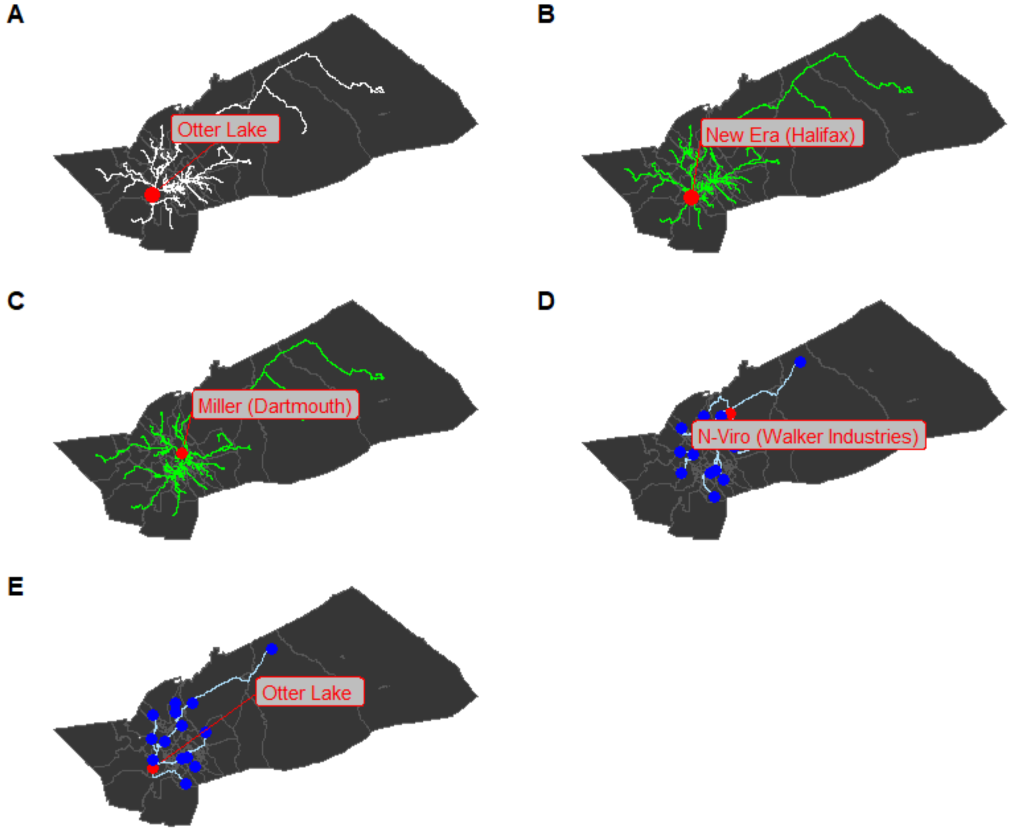

Figure 3-10 Calculated routes for different streams collection from source and transportation to the facility. Plot A is showing garbage collection from 92 districts to Otter Lake facility (white lines). Plot B and C are indicating the source separated organics collection from 92 districts (green lines) to New Era and Miller composting respectively. Plot D is showing the routes from each of 14 wastewater treatment facilities to the biosolids processing facility (N-Viro) (blue lines). Plot E is showing transportation of sludge from WWTF to Otter Lake (novel biorefinery).

3.4.2 Electricity Mix

Nova Scotia electricity generation by fuel type in 2018 is considered for the local electricity mix (Table 3-43) (Canada Energy Regulator, 2022). Ecoinvent V3 database is used for each fuel resource back ground processes.

Table 3-43 Nova Scotia Energy Mix by Fuel Type in 2019 (Canada Energy Regulator, 2022)

Fuel Type	Share (%)	Ecoinvent Process
Coal and Coke	52	electricity production, hard coal | electricity, high voltage | Cutoff, U - CA-NS

Wind	11	electricity production, wind, 1-3MW turbine, onshore | electricity, high voltage | Cutoff, U - CA-NS
Natural Gas	22	electricity production, natural gas, conventional power plant | electricity, high voltage | Cutoff, U - CA-NS
Hydro / Wave / Tidal	10	electricity production, hydro, reservoir, non-alpine region | electricity, high voltage | Cutoff, U - CA-NS
Petroleum	2	electricity production, oil | electricity, high voltage | Cutoff, U - CA-NS
Biomass / Geothermal	3	heat and power co-generation, biogas, gas engine | electricity, high voltage | Cutoff, U - CA-NS

3.4.3 Wastewater Treatment Facility

Ecoinvent V3 wastewater treatment (WWT) process (Jungbluth et al., 2007),” market for wastewater, average | wastewater, average | Cutoff, U – RoW”, is used as the background system to treat generated leachate in landfilling and composting facilities as well as excess effluent from biorefinery. The selected process mainly focuses on the technologies and processes used in Quebec. This was chosen to represent Canadian local inventory as much as it is possible. Different stages of WWT including infrastructure constructure, and chemical and energy demand of the facility is considered in the selected process. Similar characteristics for all effluents is considered in this study due to time and resource limitations. A summary of inventory used for WWT process is represented in Appendix A.

3.5 Life Cycle Impact Assessment

The results of each process life cycle inventory were entered to the openLCA software. In the next step, life cycle impact assessment is conducted to translate achieved inventory into different environmental impact categories. ReCiPe 2016 impact assessment method was used for this step. In ReCiPe methodology, two midpoint and endpoint categories are typically used. Endpoint and midpoint are indicating how selected environmental indicators are close to the end of the environmental mechanism. The ReCiPe midpoint is chosen for this analysis to cover impacts over 100 years (Bradley et al., 2022).

Seven different impact indicators are selected based on the most relevant environmental burdens to the life cycle inventory of this study: Fossil Resource Scarcity (FRS), Freshwater Eutrophication (FE), Global Warming Potential (GWP), land use, Marine Eutrophication, Terrestrial Acidification (TA), and water consumption. A summary of each indicator based on ReCiPe document (National Institute for Public Health and the Environment, 2017) is provided in the following.

Fossil Resource Scarcity (FRS), in kg oil eq, evaluates an increase in demand, or change in the future fuel production technique, source, and/or location as a result of consuming the current available fossil fuel. To calculate the related amount of alterative fossil fuel resource, ReCiPe methodology is using fossil fuel potential, which is ratio of higher heating value of the fossil fuel consumed to the crude oil. In Freshwater water Eutrophication (FE) indicator (in kg P eq), the environmental impact caused by nutrient discharge (i.e. phosphorus and nitrogen) into soil or freshwater bodies is evaluated. Increased nutrient levels may result in an increase in autotrophic and heterotrophic species nutrient uptake, which could result in the loss of related species. ReCiPe midpoint measures the intensity of P emissions to freshwater, and agriculture soil (that potentially transfer to surface water) as well as considering relative residence of P in each water body.

In Global Warming Potential (GWP) (in kg CO_2 eq), the impact of gaseous emissions on ecosystem and human health is evaluated. The rise in atmospheric concentration of greenhouse gases leads to an increase in radiative forcing capacity, which ultimately raises the average world temperature. This is the process by which gases emissions cause an adverse influence on human health and ecosystem. In GWP, ReCiPe utilizes different weighting factors to convert different gaseous impact to CO_2 equivalent. These factors are calculated by considering each gas radiative efficiency (W m^{-2}/ppb), specific mass to concentration factor (ppb/kg), lifetime of the substance and time horizon of the assessment (up to 1000 years).

Land use impact category (in m^2 crop eq.year) measures change of land cover or actual use of land as a result of different activities. This changes potentially can cause loss of habitat or soil disturbance that can damage ecosystem. The land use indicator is quantified in ReCiPe midpoint by measuring species loss due to land transformation relative to species loss because of annual crop production. The species loss itself is related to species richness alteration associated to each type of land occupation. A positive value means the land transformation is causing a decrease in species richness compared to annual crop production while a negative number shows an increase in species density as a result of the change.

Marine Eutrophication (ME) (in kg N eq) measures the environmental burden due to discharge of nutrient into marine ecosystem. This discharge can potentially cause anoxic zones due to oxygen depletion that might lead to a disturbance in marine ecosystem. N is chosen in this indicator unit over P (unlike freshwater eutrophication) since it is assumed that N is the limiting nutrient in the described mechanism. ReCiPe midpoint measures amount of equivalent N transformation to coastal, river, and watershed bdoeis from different sources as well as considering associated exposure factor of transferred N to marine ecosystem.

In Terrestrial Acidification impact (TA) (in kg SO2 eq), the alteration in soil acidity as a result of atmospheric deposition of inorganic substances is measured. This mainly affects the plant species which have a lower desired level of acidity which eventually can damage ecosystem. Major gaseous emissions that cause acidification are NO_x, ammonia and SO_2. In ReCiPe midpoint methodology, acidification impact of different emission are calculated by considering changes in air emissions and acid deposition based on climatic factors (e.g. wind speed) and calculating the change in H^+ concentration is different level of soil based on conducted mass balance.

Finally, water consumption or water use indicator (in m^3 water) evaluates the change in use of water in the life cycle of a process. Different mechanisms might cause alteration in water use, including but not limited to water evaporation, water transfer into products, disposed of water in watersheds or sea. All these changes might deprive human or ecosystem of water from its origin which can appear as water shortage for irrigation, reduction in plant diversity and change in water discharged to rivers (that affect freshwater fish species).

Chapter 4: Results and Discussion

After passing inventory analysis to openLCA and setting the impact assessment methodology (ReCiPe 2016 midpoint) in openLCA, the results associated with each product system were extracted. The results are generally divided into two main scenarios: HRM scenario and Novel Biorefinery (NB) scenario. Each sub-process for these scenarios are properly labelled to better categorize and differentiate impact of each facility. HRM Scenario subprocesses include a) HRM, BPF, b) HRM, Landfilling, c) HRM, composting, and d) HRM, Transportation. Defined subprocesses for NB scenario are a) NB, Biorefinery which is whole proposed biorefinery including the digester, CHP, and digestate valorization systems, b) NB, Landfilling, c) NB, Composting, and d) NB, Transportation. The results of each impact category is then extracted from openLCA and discussed in detail in this chapter. **Table 4-1** is representing achieved results for HRM and NB scenario and their associated main product systems while **Figure 4-1** is showing relative results plot for each impact indicator. It should be noted that "total" results for HRM and NB scenarios are calculated by summation of each sub-process in each indicator.

The total results of HRM and NB scenarios is used to compare their environmental performance. Relative results (**Figure 4-1**) is generated by scaling the total HRM and NB values in each impact category to range of -100 to 100. As it can be seen in **Table 4-1** and **Figure 4-1**, NB scenario shows lower environmental burdens in all evaluated categories. The negative numbers associated with the NB scenario is indicating environmental savings by substituting background processes with generated by-products and/or energy. A major emissions reduction is observed in land use, fossil resource scarcity and global warming potential with 922, 664, and 633% reduction, respectively. A relatively less saving was achieved in terrestrial acidification (477% reduction) and freshwater eutrophication (464% reduction), followed by marine eutrophication (33% reduction).

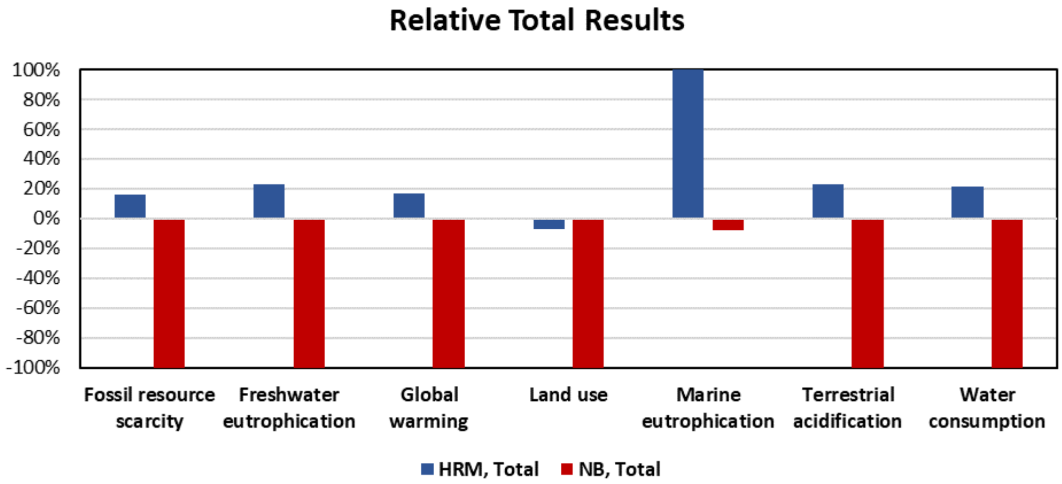

Figure 4-1 Relative total results of impact categories for HRM and NB scenarios. The total results are scaled to range of -100 to 100 percent.

Table 4-1 Overall results of each environmental impact category for each system and total scenarios

Product Systems	**Fossil resource scarcity**	**Freshwater eutrophication**	**Global warming**	**Land use**	**Marine eutrophication**	**Terrestrial acidification**	**Water consumption**
	kg oil eq	kg P eq	kg CO_2 eq	m^2a crop eq	kg N eq	kg SO_2 eq	m3
HRM Scenario							
HRM, Total	4.26E+06	9.80E+03	1.64E+07	-3.22E+03	4.26E+04	1.45E+05	2.38E+07
HRM, Landfilling	1.66E+06	8.91E+03	6.79E+06	4.25E+03	4.25E+04	4.33E+04	2.24E+07
HRM, Collection	8.96E+05	7.60E+01	2.71E+06	7.55E+03	1.57E+01	9.69E+03	7.99E+05
HRM, BPF	7.54E+05	-1.76E+02	1.89E+06	-2.59E+04	-1.33E+01	4.45E+03	1.39E+06
HRM, Composting	6.79E+05	9.23E+02	4.25E+06	-2.04E+04	1.19E+02	8.45E+04	-1.42E+06
HRM, Distribution	2.75E+05	6.49E+01	7.71E+05	3.13E+04	7.45E+00	2.55E+03	5.99E+05
NB Scenario							
NB, Total	-2.40E+07	-3.57E+04	-8.74E+07	-3.29E+04	2.87E+04	-5.44E+05	-9.24E+07
NB, Landfilling	1.22E+06	6.57E+03	5.51E+06	3.97E+03	3.20E+04	3.00E+04	1.66E+07
NB, Collection	1.07E+06	9.05E+01	3.22E+06	8.99E+03	1.86E+01	1.15E+04	9.51E+05
NB, Composting	3.12E+05	4.54E+02	2.11E+06	-1.06E+04	-1.01E+01	4.22E+04	6.62E+05
NB, Distribution	9.31E+04	2.20E+01	2.61E+05	1.06E+04	2.53E+00	8.65E+02	2.03E+05
NB, Biorefinery[a]	-2.67E+07	-4.28E+04	-9.85E+07	-4.58E+04	-3.27E+03	-6.29E+05	-1.11E+08
Total reduction	664%	464%	633%	922%	33%	477%	489%
Landfill reduction	26%	26%	19%	7%	25%	31%	26%
Compost reduction	54%	51%	50%	48%	108%	50%	147%
Transportation reduction	0.93%	20.18%	-0.18%	49.55%	8.41%	-1.30%	17.41%

[a] NB, Biorefinery is representing the proposed biorefinery system including digesters, CHP unit, and digestate valorization system.

To further assess the impact of each sub-process on total results, a breakdown of different contributors in each impact category is represented in the rest of this section. 8 main product systems are considered for the detailed analysis: landfilling (in both HRM and NB scenarios), composting (in both HRM and NB scenarios), transportation (in both HRM and NB scenarios), biorefinery (only in NB scenario) and biosolid processing facility (BPF) (only in HRM scenario). Each of the mentioned product systems also consist of connected background processes (e.g., electricity generation//consumption, heat generation/consumption, fertilizer substitution, etc.). Cut-off method based on ISO 14040 (ISO, 2006) is used to filter the most contributing processes in each impact category. Cut-off criteria for this study was considered 1% of total NB or HRM value, whichever that has higher absolute value. The cut-off criteria is different for each indicator and was separately calculated.

4.1 Fossil Resource Scarcity

The Fossil Resource Scarcity (FRS) is reduced from 4.26E6 kg oil_{eq}/ FU in HRM scenario to -2.40 E7 kg oil_{eq}/FU in NB scenario, equal to 664% reduction. The most contributing product system in FRS is the biorefinery system with an environmental saving equal to -2.67E07 kg oil_{eq}/ FU. As it is shown in **Figure 4-2**, excess generation of electricity from CHP unit plays the most critical role in the biorefinery product system (-2.5E7 E7 kg oil_{eq}/FU). After electricity, generated ammonium sulfate (AmS) (-6.9E5 kg oil_{eq}/FU), excess heat from CHP (-6.5 E5 kg oil_{eq}/FU), and struvite N substitution by generated struvite (-4E5 kg oil_{eq}/FU) had the most environmental savings. The impact of fertilizer substitution (N, P and K) from generated compost and Halifax soil amendment are eliminated after cut-off method implementation.

Landfilling facilities, biosolid processing facility (BPF), and composting sub-systems are identified as major contributors to FRS indicator with positive value (increasing environmental burdens). Wastewater treatment (WWT) activities for leachate treatment and electricity consumption are identified as the main impactful factors in landfilling facilities (54 and 31% of total HRM, landfilling and 55 and 17% of total NB, landfilling, respectively). Diversion of waste from landfills to biorefinery (that leads to less leachate generation), and elimination of stabilization unit (that reduces the electricity requirement) reduced total landfilling FRS impact by 26%. For BPF facility, required heat is identified as the background process with a major environmental burden with 104% of the total impact (the sub-process of heat consumption is greater than the overall BPF value since the sub-process of fertilizer replacement lowers the overall impact, hence the percentage is higher than 100). The BPF impact and its heat consumption is completely removed in the NB scenario by diversion of sludge to the digester. In composting product system, electricity consumption has the highest environmental burden in FRS category (114% of overall composting value). Diversion of SSO from composting to biorefinery in the NB scenario and replacing composting facility feed with already digested waste reduced the burden in composting by 54%.

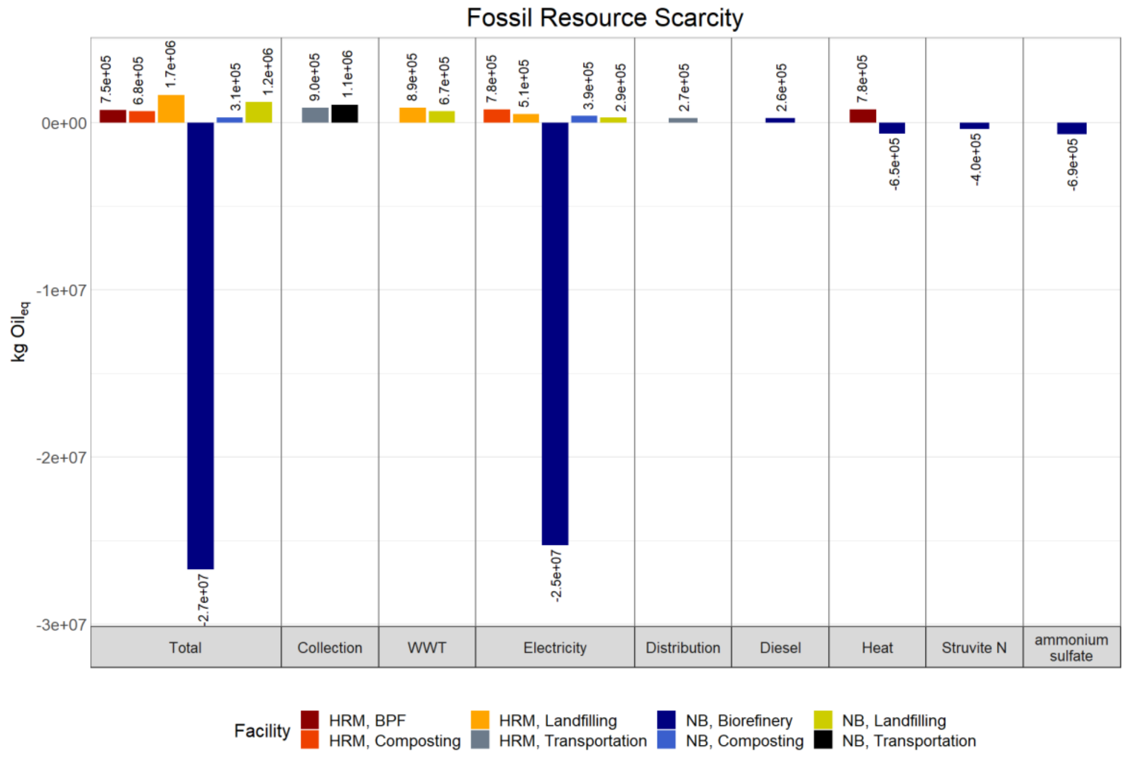

Figure 4-2 Fossil Resource Scarcity (FRS) impact category results for main product systems.

In general, the proposed biorefinery was able to reduce the impact by a) generating value-added products and energy, and b) diverting a portion of waste from conventional facilities (composting, landfilling, and BPF) that reduces the WWT activities emissions and heat/electricity requirements. A similar trend is observed in Herrera et al. study, where organic waste digester coupled with ammonium sulfate production is compared against mineral fertilizer production (Herrera et al., 2022). According to their results, the AD system was able to reduce the FRS impact by 168% (from 329 to -224 kg oil_{eq}/ FU) with electricity generation as the main driving factor (-384 kg oil_{eq}/ FU). Weligama Thuppahige & Babel also investigated the environmental impacts of anaerobic digestion for the organic fraction of municipal solid waste with a functional unit of treating 1 t of waste (Weligama Thuppahige & Babel, 2022). Their result showed a total -0.158 kg oileq/FU for FRS indicator mainly driven by electricity generation (-0.161 kg oil_{eq}/ FU). A higher reduction is observed in this study mainly due to the NS electricity mix which has higher overall emission.

4.2 Freshwater Eutrophication

In Freshwater Eutrophication (FE), a total 464% reduction (9.8E3 to -3.57E4 kg P_{eq}/FU) is observed from HRM to NB scenario. The composting and landfilling facilities in both scenarios and biorefinery from the NB scenario are identified as major contributors to the FE impact category. The proposed biorefinery product system showed a negative value (environmental saving) equal to -4.28E4 kg P_{eq}/FU. Electricity and heat generation from CHP followed by AmS production are identified as major contributors to this saving. As was discussed in the inventory analysis section, 52% of Nova Scotia electricity mix is generated from coal as the fuel source. The major environmental saving in the FE category from electricity generation is related to the substitution of hard coal mine operation and hard coal preparation processes.

The positive value for landfilling is mainly due to the WWT process to treat leachate. This mainly originated from the facility infrastructure (construction of the facility itself and associated sewer grid) as well as the consumption of chemicals (e.g., iron (III) chloride) in the WWT plant. In addition to WWT, electricity consumption in composting and landfilling facilities is another process that contributes a positive impact.

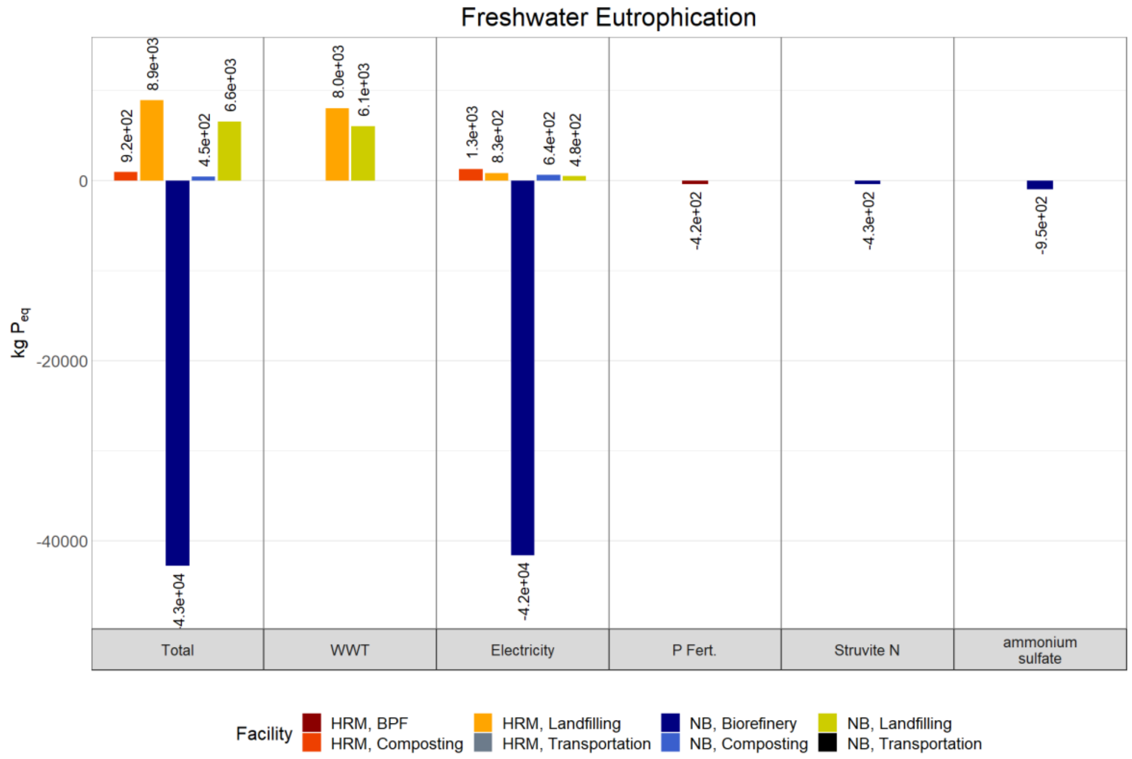

Figure 4-3 Freshwater Eutrophication (FE) impact category results for main product systems.

NB scenario showed a better performance in the FE category due to excess generation of energy from the CHP plant as well as diverting a portion of waste from composting and landfilling facilities. Weligama Thuppahige & Babel reported a positive total value (2.92E-3 kg P_{eq}/FU) for the anaerobic digestion unit, mainly due to the direct discharge of N and P from liquid digestate (Weligama Thuppahige & Babel, 2022). Herrera et al also identified energy generation as the main factor for decreasing environmental burden in FE indicator when the fertilizer recovery was conducted from digestate (Herrera et al., 2022). The results of this study for FE are also in line with the reported values showing effectiveness of the digestate valorization system for diverting nutrients from being discharged directly and major environmental savings from electricity generation.

4.3 Global Warming Potential

With regard to Global Warming Potential (GWP), a 633% reduction is identified from HRM scenario to NB scenario (from 1.64E+07 to -8.74E+07 kg $CO_{2,eq}$ /FU). Among main contributing product systems, landfilling (both scenarios), composting (both scenarios), transportation (both scenarios), BPF facility (HRM scenario) and biorefinery (NB scenario) can be mentioned. Electricity generation in biorefinery with -1.0E8 kg $CO_{2,eq}$ /FU has the highest environmental saving in GWP. After electricity, AmS production with -2.3E6 kg $CO_{2,eq}$ /FU and heat generation with -1.9E6 kg $CO_{2,eq}$ /FU are identified as a major environmental savings.

In terms of the product systems with a positive impact, landfilling facilities with 6.8E6 and 5.5E6 kg CO2,eq /FU for HRM and NB scenarios, respectively, had the highest values. This is mainly coming from WWT activities to treat leachates followed by electricity requirements. Composting facilities also showed a positive overall impact mainly due to electricity associated with the process. For HRM composting, the direct emission (2.3E6 kg CO2,eq /FU) is also identified as one of the contributors. The direct emission is mainly due to unwanted methane generation (1.7E6 kg $CO_{2,eq}$ /FU) and dinitrogen monoxide (5.0E5 kg $CO_{2,eq}$ /FU) emissions. A lower direct emission was observed in NB composting. This is because the SSO stream is diverted to the biorefinery and only solid digestate is sent to the composting facilities in NB scenario. As a result of these changes in the landfilling and composting facilities, the overall result of these facilities is reduced by 19 and 50%, respectively, from HRM to NB scenario. No significant change is observed in transportation emissions between HRM and NB scenarios (0.18% change).

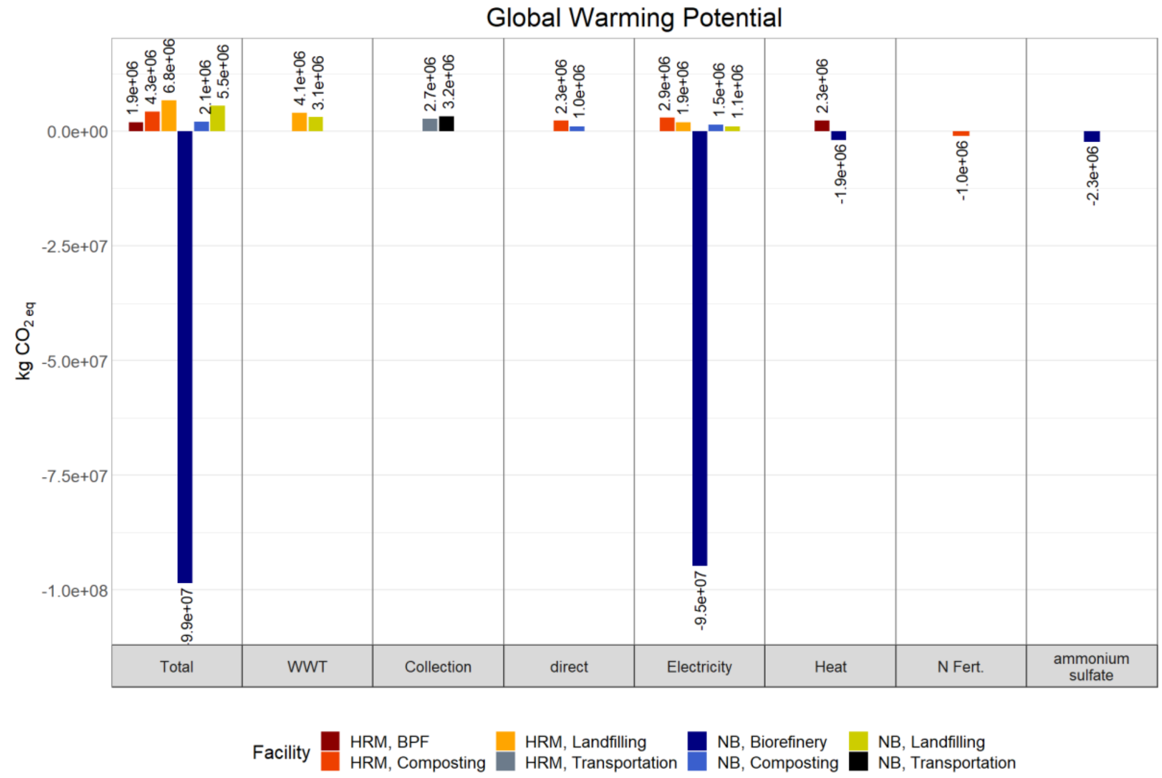

Figure 4-4 Global Warming Potential (GWP) impact category results for main product systems.

Overall, the GWP impact category was decreased in the NB scenario mainly due to the substitution of electricity and heat generation as well as AmS production. The alteration in landfilling and composting also contributed to reducing the GWP impact. Weligama Thuppahige & Babel reported an environmental burden of 232 kg $CO_{2,eq}$ /FU for anaerobic digestion of OFMSW, mainly coming from direct air emissions, specifically methane (Weligama Thuppahige & Babel, 2022). In Herrera et al. work, a positive value is identified for the GWP indicator, equal to 3354 kg $CO_{2,eq}$ /FU. This positive value was mainly due to emissions from spreading the fertilizers on the land (3999 kg $CO_{2,eq}$ /FU) while electricity generation is the main environmental saving factor (-1315 kg $CO_{2,eq}$ /FU) (Herrera et al., 2022). Since the scope of this study is not evaluating the spread of products on the soil, a lower total value is observed in the GWP indicator. Also, electricity substitution saving is higher as a result of many intense emissions associated with the NS mix. It should be also noted that direct emissions from novel biorefinery (mainly from the digester and CHP units) were eliminated due to cut-off method implementation. This is because much higher numbers associated with savings (especially from electricity) significantly affected the total value and increase the cut-off threshold.

4.4 Land Use

In terms of land use, a 992% decrease is identified from the HRM scenario to the NB scenario (from -3.22E3 to -3.29E4 m^2 Crop eq/FU). An unexpected negative (land saving) value is observed in the land use indicator of electricity generation for the Nova Scotia mix (negative for the HRM scenario and positive for the NB scenario). This is due to background coal production and operation process, which includes the occupation of land for dumping the mined coal. The occupation land for dumping in the ReCiPe impact assessment method has a negative weight factor. This means that upon using the coal, in this case generating electricity, the dumping site will get emptier. As a result, generating electricity shows a total negative land use. This negative value for electricity generation can be seen in composting and landfilling facilities. Moreover, the production of fertilizer in BPF, composting and biorefinery showed negative land use impact as well. This is due to the elimination of conventional fertilizer production facilities. In biorefinery, the elimination of electricity generation resulted in a positive value. This positive value is balanced with negative values from other by-products.

Among other negative values (processes that reduce the impact), biorefinery struvite production showed the highest contribution with -2E5 and -1E5 m^2 Crop eq/FU. Direct impact from biorefinery, transportation, electricity generation in biorefinery, and landfill leachate treatment in WWT are among the major processes with positive value (adverse environmental impact). The direct impact of the biorefinery is reflecting the required land for algae cultivation, equal to 6.9 ha.

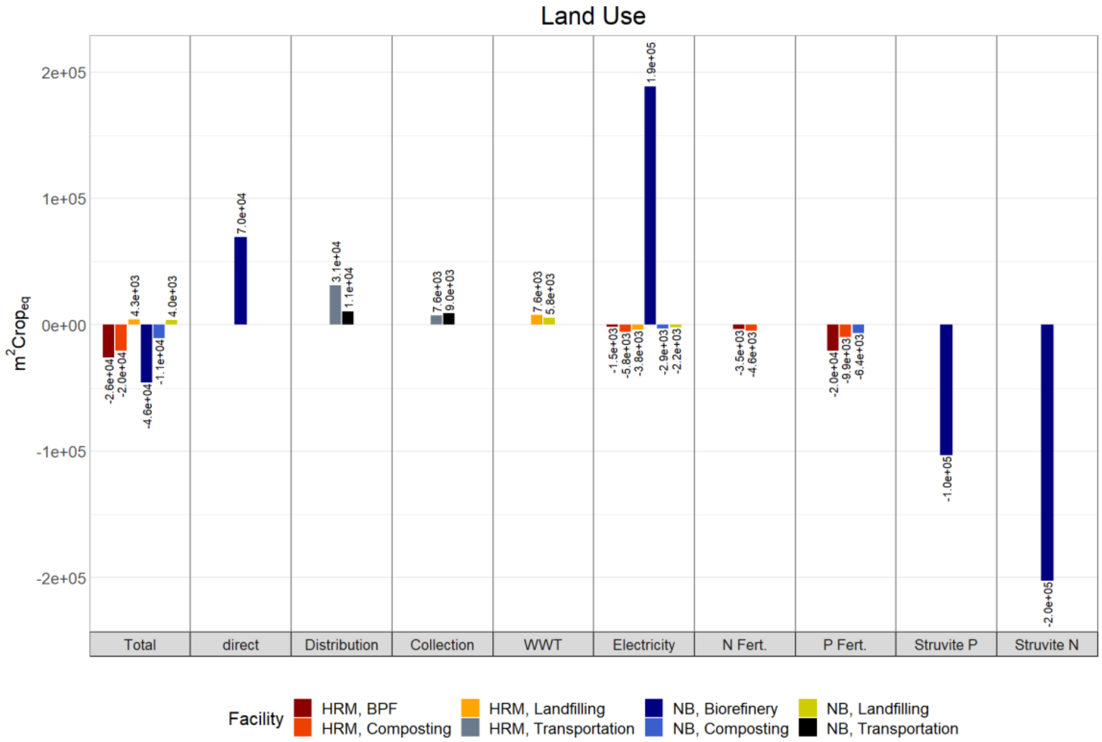

Figure 4-5 Detailed analysis of Land Use impact category for HRM and NB scenarios.

In general, results are suggesting that the NB scenario is lowering the conversion of land compared to the HRM scenario. Herrera et al. suggested a net positive value for land use for fertilizer recovery from AD (Herrera et al., 2022). They cited the production of renewable energy as the key source of savings, whereas the AD plant, digestate dissemination, and transportation as the main sources of increasing land use burden. The results of this study are showing different results mainly due to the positive value that can be seen in the electricity generation that substituting coal mine operation.

4.5 Marine Eutrophication

In Marine Eutrophication (ME) impact category, the environmental burden of the HRM scenario is reduced by 33% (from 4.26E4 to 2.87E4 kg N eq/FU, for HRM and NB scenarios respectively). Landfilling facilities (for both NB and HRM scenarios) and biorefinery (NB scenario) are identified as major contributing product systems. High values for landfilling facilities are generated from the leachate treatment in WWT mainly due to released emissions associated with the infrastructure requirements. A negative value for biorefinery is identified mainly as a result of electricity and soybean production. The negative value associated with electricity is due to avoiding coal mine operation and hard coal operation which saves emissions from spoil from hard coal mining treatment. For soybean production, which is substituted by algae production, environmental savings can be seen from avoiding soybean sowing activities (soybean label refers to eliminated soybean process as a result of algae production).

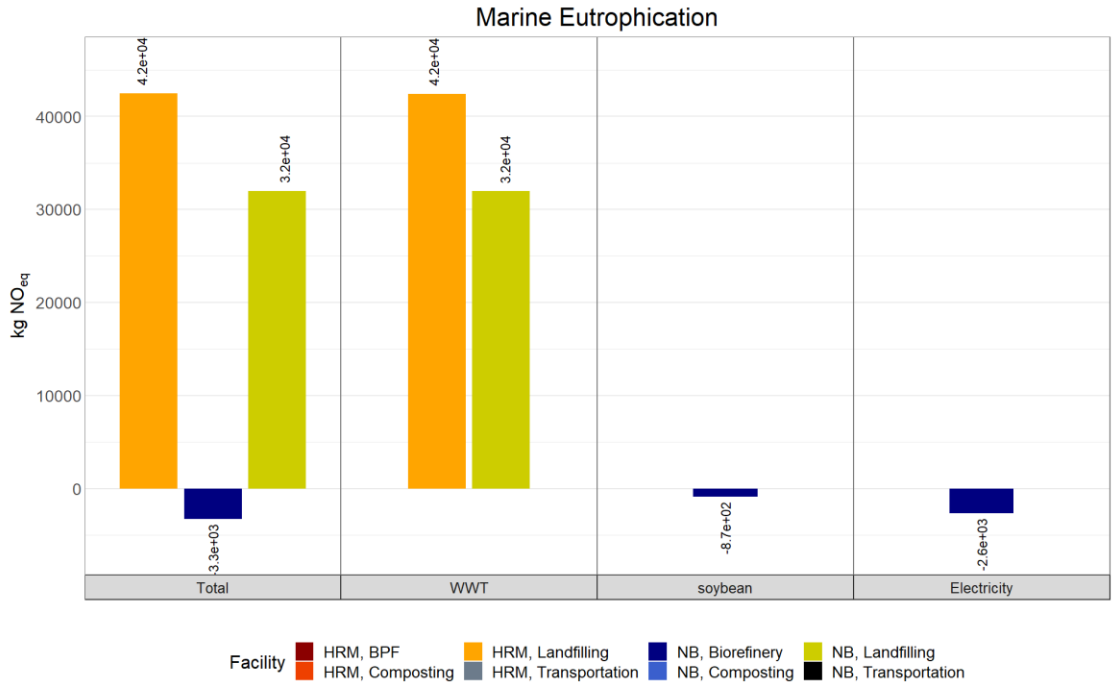

Figure 4-6 Detailed analysis of marine eutrophication impact category for HRM and NB scenarios.

The result of the study aligns with the majority of the literature, suggesting electricity generation as the main saving. Weligama Thuppahige & Babel reported a negative ME for AD unit (-5.72E6 kg N eq/FU, mainly due to generated electricity savings (-6.13 E6 kg N eq/FU) (Weligama Thuppahige & Babel, 2022). Other studies indicated negligible savings from electricity for AD units, mainly impacted by other positive processes including digestate land application and different electricity mixes (Guven et al., 2019; Slorach et al., 2019). Among the mentioned reason, the electricity mix likely has a higher contribution to the observed difference between this study and others.

4.6 Terrestrial Acidification

A total 548% saving is observed in Terrestrial Acidification (TA) category, reducing the HRM scenario impact from 1.45E5 to -5.44E5 kg SO_2 eq/FU in the NB scenario (477% reduction). Excess electricity, struvite, and ammonium sulfate (AmS) production from biorefinery are identified as major processes with environmental savings (negative values). Avoiding nitrogen oxides and sulfur dioxide because of the elimination of generating electricity from hard coal yields a significant saving for electricity generation in biorefinery. For AmS, avoided emissions are mainly due to avoided heat and organic chemical factory construction emissions that are used in the production of conventional AmS. For Struvite as nitrogen, the substitution of phosphoric acid production that is used to generate conventional struvite production is the main driver of environmental savings.

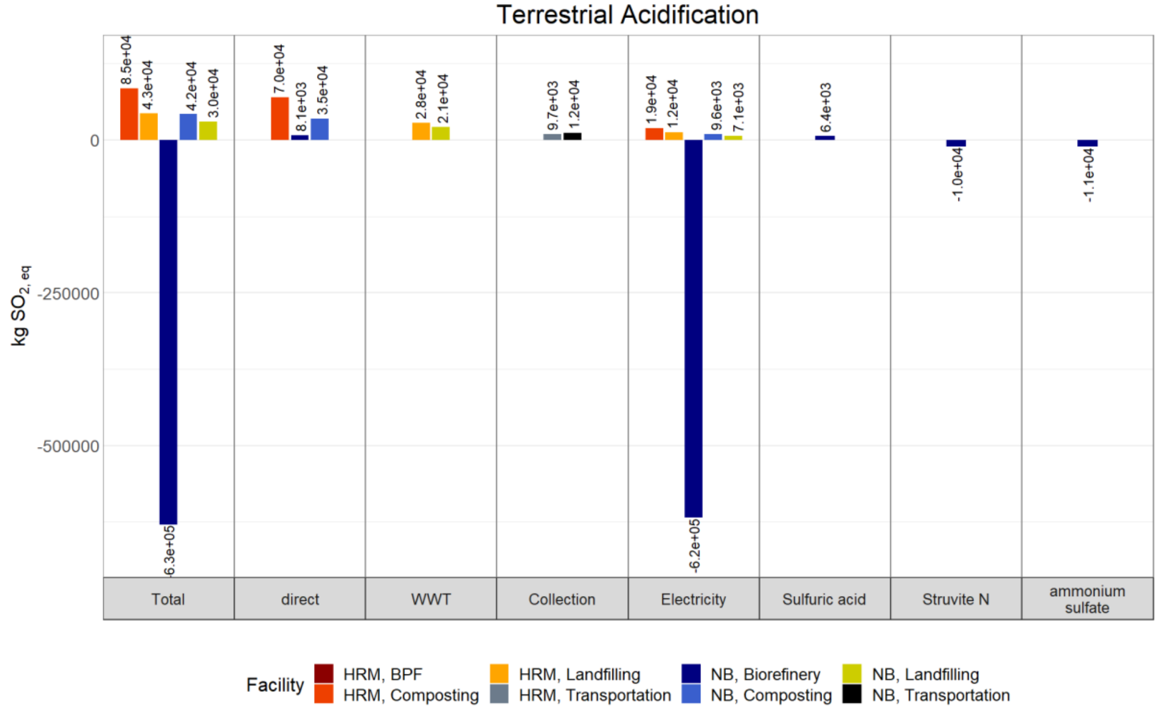

Figure 4-7 Detailed analysis of terrestrial acidification impact category for HRM and NB scenarios.

Composting, landfilling and transportation emissions are considered as the main product systems with adverse (positive) impacts in the TE category. For composting, direct emissions from each facility and electricity requirements are the main contributors. Direct emissions are representing ammonia that is released after biofiltration. As it was mentioned before, the production of hard coal is the main reason behind electricity generation emissions for composting and landfilling facilities. Wastewater treatment activities to treat generated leachate are another positive contributor in the TA category for landfilling facilities. This is because of a combination of emissions released from background activities including sewer grid, facilities construction, chemicals (e.g. iron (III) chloride, aluminum sulfate), etc.

Implementing novel biorefinery was able to reduce the TA environmental burdens of HRM waste management systems by further expanding the products (mainly electricity, AmS, and struvite) while diverting a portion of waste from composting and landfilling processes. The results of the study are in line with the literature, reporting electricity generation as the main TA indicator saving in the MSOF AD (Herrera et al., 2022; Weligama Thuppahige & Babel, 2022).

4.7 Water Use

Based on the LCA results, the total water consumption of the HRM scenario decreased by 489% when in the NB scenario (from 2.38E7 to -9.24E7 m^3/FU). Electricity, ammonium sulfate (AmS), and struvite as N fertilizer are among the main processes that yield environmental savings. Water use in the electricity section is mainly related to water consumption in the hydro process followed by hard coal resources. This water requirement is avoided when biorefinery is generating excess electricity. For AmS, a negative impact (-4.9E6 m^3/FU) is achieved. This negative impact is mostly coming from avoiding water use in the organic chemical factory which is a background process of conventional AmS production. For Struvite as N and P fertilizer substitution (-2.9E6 and -1.5E6 m^3/FU, respectively), water consumption is avoided by eliminating phosphoric acid production. The P fertilizer negative impact is a result of the elimination of single superphosphate production in the background processes related to the P fertilizer market.

Among positive processes, landfilling has a major impact on water consumption, mainly due to WWT activities to treat leachate. This is due to numerous background activities associated with the wastewater treatment plant construction and operation including energy requirements and chemical requirements.

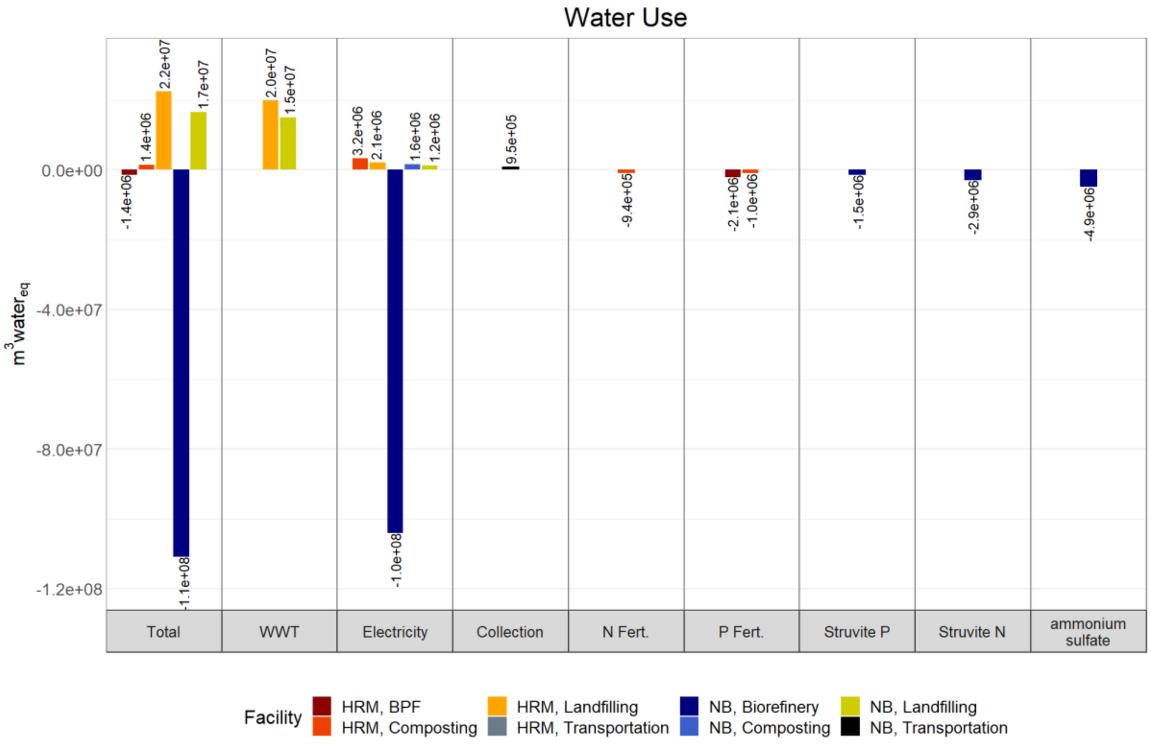

Figure 4-8 Detailed analysis of water use impact category for HRM and NB scenarios.

According to Weligama Thuppahige & Babel, a positive value for AD (adverse environmental impacts) is reported because of the required water for the digester (Weligama Thuppahige & Babel, 2022). In Herrera et al. study, water consumption is negative, mainly due to renewable electricity generated from CHP (Herrera et al., 2022). The result of this study also showed excess electricity generation as a major environmental saving, and significant water saving as a result of implementing a digestate valorization system to recycle processed water to the digesters.

4.8 Sensitivity Analysis

A sensitivity analysis is conducted to assess the robustness of the LCA model to major considered inputs and parameters. Based on the achieved results, the electricity mix plays a critical role in each evaluated impact category. As a result, other practical electricity mixes available to Nova Scotia are evaluated to see the model response to electricity source alteration. Two other electricity mixes are used: a) Quebec electricity mix and b) New Brunswick electricity mix. Both of these electricity mixes are chosen in light of the nova scotia electricity plan for 2020-2040 (NS Government, 2015). This plan suggested a potential for electricity transmission from these two provinces to NS via maritime link. The resources for electricity generation are extracted from the governmental database (Canada Energy Regulator, 2022).

Table 4-2 Summary of different electricity mixes included in sensitivity analysis

Fuel type	Baseline (NS mix)	Sensitivity 1 (QB mix)	Sensitivity 2 (NB mix)
Coal and Coke	52	-	14
Wind	11	5	7
Natural Gas	22	0.1	15
Hydro / Wave / Tidal	10	94	22
Petroleum	2	0.2	1
Biomass / Geothermal	3	0.7	4
Uranium	-	-	38

The result of the sensitivity analysis is shown in Figure 4-9. To enable comparison and visualization, the values in each impact category are standardized in the range of -100 to 100. The results plot is showing HRM and NB scenarios variation in response to three electricity mixes. Baseline, SENS1, and SENS2 cases are showing NS, Quebec, and New Brunswick electricity mixes, respectively. Overall, the electricity mix alteration did not show a significant variation in the HRM system's impact in all evaluated categories. On the other hand, a significant variation can be observed in the NB scenario, due to changes in associated emissions for biorefinery electricity generation. This shows that the results of the model for biorefinery are highly dependent on the location of the study as well as the electricity mix of the region. In all evaluated electricity mixes proposed biorefinery scenario is showing a better environmental performance (lower impact values) in all impact categories. This likely suggests that the biorefinery is going to increase the environmental savings if implemented for other regions.

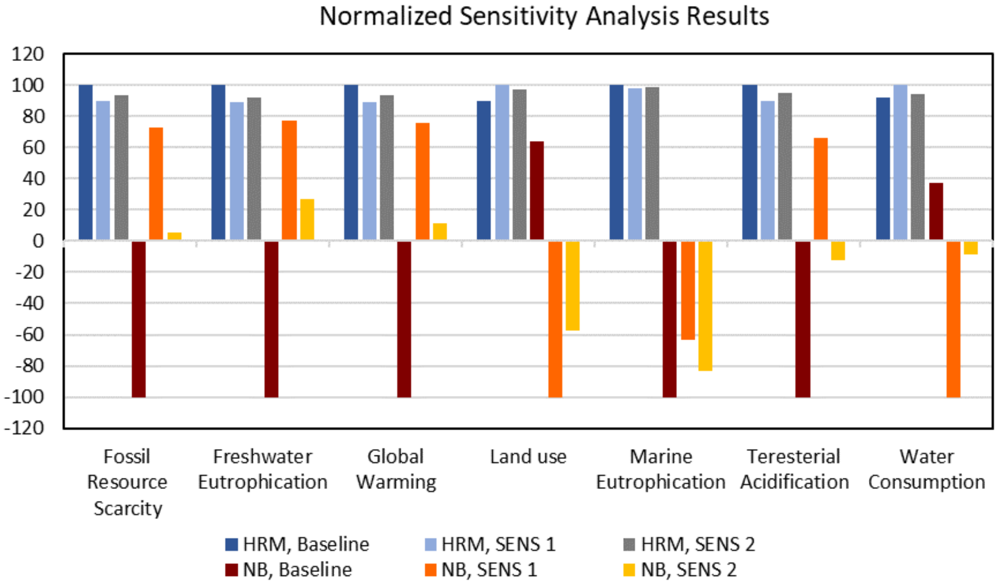

Figure 4-9 Normalized results of each impact category for each sensitivity case for HRM and Novel Biorefinery (NB) scenarios. Different sensitivity cases are result of change in electricity mix. Baseline, SENS1 and SENS2 are reflecting NS, Quebec, and New Brunswick electricity mixes.

Since the variation in electricity mix impacted the biorefinery product system more than other systems, a further breakdown is provided in this section. The total value for the proposed biorefinery system for each impact category and associated change as a result of the electricity mix is shown in Table 4-3. Figure 4-10 is also showing scaled results for the biorefinery product system in each sensitivity case. Quebec electricity mix (SENS1) showed overall lower emissions in fossil resource scarcity, freshwater eutrophication, global warming, and terrestrial acidification, followed by the New Brunswick mix (SENS2) and Nova Scotia (Baseline) mix. For the land use and water consumption categories, the SENS1 and SENS2 decreased biorefinery burdens as the Quebec and New Brunswick electricity mix have lower impact values in these categories.

Table 4-3 Biorefinery product system result for sensitivity analysis cases in different impact categories. Values are based on the study functional unit, which is generated waste in HRM (126000 t/y).

Scenario	Unit	Baseline	SENS1, Quebec mix		SENS2, New Brunswick mix	
		Value	Value	Change	Value	Change
Fossil Resource Scarcity	kg oil eq	-2.67E+07	-1.61E+06	-94	-1.14E+07	-57
Freshwater Eutrophication	kg P eq	-4.28E+04	-1.39E+03	-97	-1.32E+04	-69

Global Warming	kg CO2 eq	-9.85E+07	-4.77E+06	-95	-3.91E+07	-60
Land use	m2a crop eq	-4.58E+04	-2.38E+05	420	-1.88E+05	310
Marine Eutrophication	kg N eq	-3.27E+03	-6.53E+02	-80	-2.07E+03	-37
Teresterial Acidification	kg SO2 eq	-6.29E+05	-4.04E+04	-94	-3.17E+05	-50
Water Consumption	m3	-1.11E+08	-4.15E+08	274	-2.13E+08	92

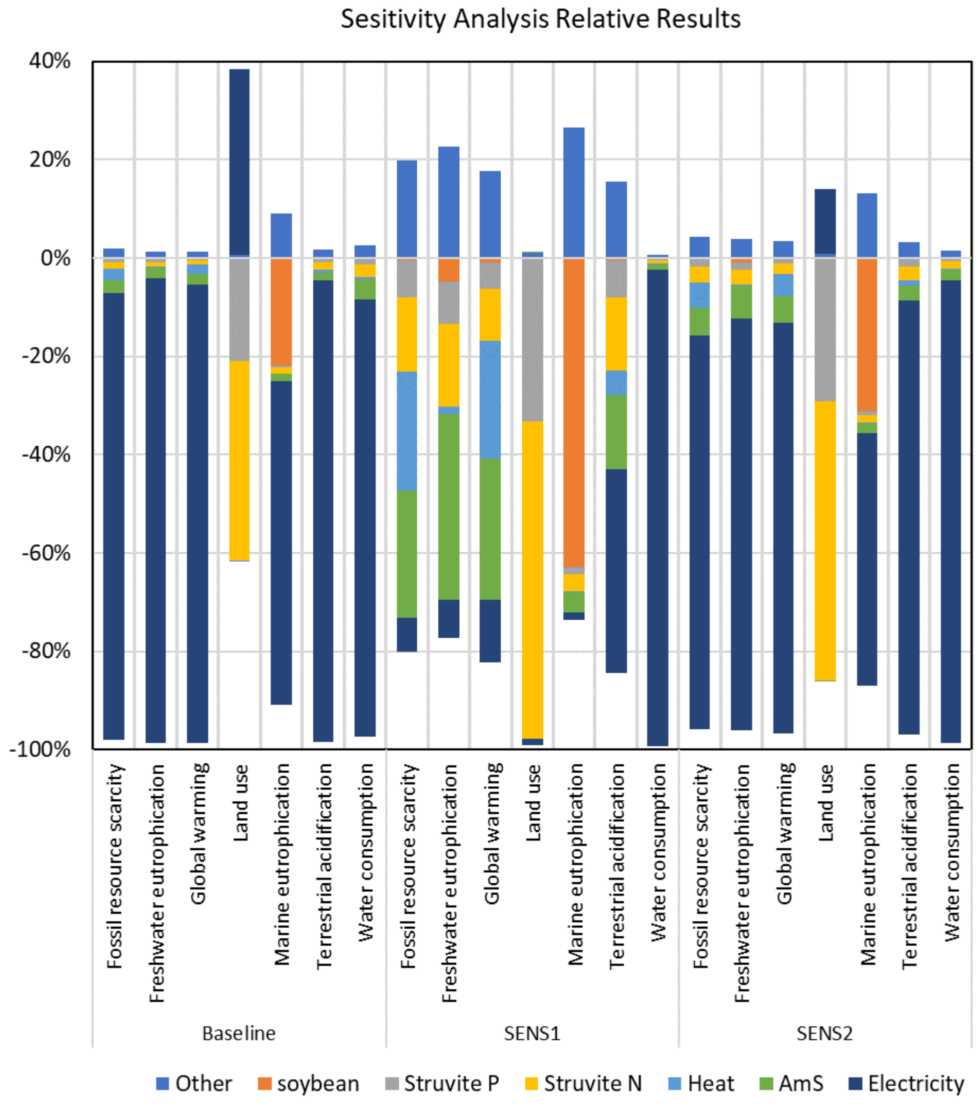

Figure 4-10 Scaled results of biorefinery product system for each impact category for each sensitivity analysis case. “Other” processes include processes with positive impact (e.g. chemicals requirement, construction, etc.). It was observed that the electricity alteration changes other by-products contribution as well.

It is also observed that the electricity mix variation changes other by-products contribution in biorefinery product system. As it is shown in Figure 4-10, environmental savings from electricity generation is dominant in Baseline case. This is mainly due to high emissions associated to electricity generation from coal, which is avoided by generated electricity from biorefinery. In the SENS1, electricity generation mix contributes to a relatively lower savings, as Quebec electricity is “cleaner” than NS electricity mix. As a result of reduction in environmental savings by electricity generation, other by-products and processes relative impact becomes more important in total environmental performance of the system. This can be seen in increased contribution of ammonium sulfate, struvite, and heat processes in fossil resource scarcity, freshwater eutrophication, global warming, and terrestrial acidification categories. The changes in SENS2 is less significant compared to SENS1, as New Brunswick mix is more similar to NS mix compared to Quebec.

In general, the sensitivity analysis showed variability of the system in response to electricity mix alterations. The proposed biorefinery showed a consistent beneficial application in all evaluated electricity mixes compared to HRM conventional scenario. The results also indicated that the contribution of other by-products, including struvite, ammonium sulfate, and algae can be increased if a cleaner electricity mix is used.

Chapter 5: Conclusions

This study conducted a life cycle assessment analysis on a proposed biorefinery for solid organic waste, focusing on maximizing nutrient and energy recovery. The novel biorefinery implemented anaerobic digestion along with struvite, ammonia stripping, and algae cultivation units. The biorefinery was compared with HRM's conventional waste management system to identify the extent of environmental burdens associated with each system. Throughout the life cycle inventory analysis, the site-specific information was used for the conventional system and supplemented with literature values when it was needed. The life cycle inventory results were analyzed via openLCA software to achieve environmental burdens in seven different impact categories.

The LCA result showed environmental savings in response to the implementation of the proposed biorefinery, compared to HRM conventional system. Electricity generation was identified as a major contributor to reducing environmental burdens to a great extent in most of the indicators. After electricity, other generated by-products including ammonium sulfate and struvite showed effectiveness in reducing each impact. Algae cultivation, which substitutes soybean cultivation for animal feed, showed the lowest savings. This was primarily caused by large energy generation environmental savings, which decreased the contribution of other by-products, especially generated algae. This hypothesis was confirmed by conducting a sensitivity analysis on the electricity mix which showed a higher contribution of all by-products when a cleaner electricity mix is substituted by generated electricity.

In addition to direct savings from by-products, implemented digestate valorization achieved lower water consumption and eutrophication impacts compared to conventional AD systems because of processing liquid digestate. This treatment enabled the option of recycling processed water to the digester directly reducing the water requirements as well as reducing the facility effluent that reduces the WWT activities emissions. Considering the positive results achieved by the novel biorefinery, a set of feature works is also identified to further check the feasibility of the proposed process. A pilot scale unit is required to further evaluate the process parameters and assess the composition of the product and influents.

Emerging contaminants (ECs) are among other measurements that should be evaluated to ensure the feasibility of the proposed system in treating water and potentially replacing conventional wastewater treatment for liquid digestate. Emerging contaminants are a group of chemical compounds that are not commonly monitored in wastewater treatment plants while they are proven or suspected to have adverse environmental and health impacts (Rodriguez-Narvaez et al., 2017). Among emerging contaminants, pharmaceuticals and personal care products (PPCPs) and endocrine disrupting compounds (EDCs) can be mentioned (Bolong et al., 2009). It is reported that the majority of emerging contaminants cannot be degraded in an anaerobic digester which poses several operational/environmental risks for downstream processes, including any digestate valorization system (Malhotra et al., 2022). The proposed algae cultivation unit can potentially reduce the digestate emerging pollutant concentration (Lamolinara et al., 2022; Tawfik et al., 2022), however, a thorough pilot scale study is required to study the effluent properties of the biorefinery.

The application of strong acids, such as sulfuric acid which is used in the ammonia scrubber unit, can potentially lead to safety and environmental issues. In addition, it can cause corrosion to the equipment in long term, making their application more limited (Jamaludin et al., 2018). Moreover, a detailed techno-economic analysis is also required to ensure the feasibility of the system. Among other potential challenges, digestibility issues of the microalgae as animal feed can be also mentioned which requires further experiments. Moreover, biomass productivity can be also further investigated which can potentially lead to more algae production and more environmental savings associated with that.

During the preparation of this document, HRM announced the plan to deactivate the front-end sorting and waste stabilization unit in the Otterlake landfill facility (Preston, 2022). This plan can potentially increase landfill emissions by not stabilizing organics and no mechanical pre-treatment. Also, this plan might impact the proposed biorefinery as Otterlake no longer generates mechanically separated organic fraction (MSOF). However, the proposed biorefinery can use existing mechanical pre-treatment equipment and separate organics to produce biogas in the proposed AD system. This potentially can mitigate potential long-term adverse environmental impacts from the Otterlake facility.

References

Aim Environmental Group. (2017). *Halifax Regional Municipality Ragged Lake Source Separated Organics Composting Facility*. https://aimgroup.ca/wp-content/uploads/2019/10/Final-as-presented-November-23-17-Ragged-Lake-CLC-Meeting-1-Presentation.pdf

Aim Environmental Group. (2019). *Ragged Lake Source Separated Organics Compost Facility Monthly Report*. https://aimgroup.ca/wp-content/uploads/2020/02/Monthly-Report-December-2019.pdf

Aim Environmental Group. (2020). *Halifax Regional Municipality Ragged Lake Source Separated Organics Composting Facility*.

Ajmal, M., Aiping, S., Uddin, S., Awais, M., Faheem, M., Ye, L., Rehman, K. U., Ullah, M. S., & Shi, Y. (2020). A review on mathematical modeling of in-vessel composting process and energy balance. *Biomass Conversion and Biorefinery*, 1–13. https://doi.org/10.1007/S13399-020-00883-Y/FIGURES/5

Alexander, A., Burklin, C., & Singleton, A. (2005). *Landfill Gas Emissions Model (LandGEM) Version 3.02 User's Guide*. US EPA.

Alfonsín, C., Hernández, J., Omil, F., Prado, Ó. J., Gabriel, D., Feijoo, G., & Moreira, M. T. (2013). Environmental assessment of different biofilters for the treatment of gaseous streams. *Journal of Environmental Management*, *129*, 463–470. https://doi.org/10.1016/J.JENVMAN.2013.08.009

Allen, D. T., & Shonnard, David. (2012). *Sustainable engineering : concepts, design, and case studies* (1st edition). Prentice Hall.

Allesch, A., & Brunner, P. H. (2014). Assessment methods for solid waste management: A literature review. *Waste Management and Research*, *32*(6), 461–473. https://doi.org/10.1177/0734242X14535653

Andreoli, C. V., Sperling, M. von, & Fernandes, F. (2017). *Sludge Treatment and Disposal, Volume 6*. IWA Publishing. https://app.knovel.com/hotlink/toc/id:kpSTDV0007/sludge-treatment-disposal/sludge-treatment-disposal

Antakyali, D., Meyer, C., Preyl, V., Maier, W., & Steinmetz, H. (2013). Large-scale application of nutrient recovery from digested sludge as struvite. *Water Practice and Technology*, *8*(2), 256–262. https://doi.org/10.2166/WPT.2013.027

Appels, L., Baeyens, J., Degreve, J., & Dewil, R. (2008). Principles and potential of the anaerobic digestion of waste-activated sludge. *Progress in Energy and Combustion Science*, *34*(6), 755–781.

Arulrajah, A., Mohammadinia, A., D'Amico, A., & Horpibulsuk, S. (2017). Effect of lime kiln dust as an alternative binder in the stabilization of construction and demolition materials. *Construction and Building Materials*, *152*, 999–1007.

Asefi, H., Shahparvari, S., & Chhetri, P. (2020). Advances in sustainable integrated solid waste management systems: lessons learned over the decade 2007–2018.

Journal of Environmental Planning and Management, *63*(13), 2287–2312. https://doi.org/10.1080/09640568.2020.1714562/SUPPL_FILE/CJEP_A_1714562_SM1047.DOCX

Askri, A., Laville, P., Trémier, A., & Houot, S. (2016). Influence of Origin and Post-treatment on Greenhouse Gas Emissions After Anaerobic Digestate Application to Soil. *Waste and Biomass Valorization*, *7*(2), 293–306. https://doi.org/10.1007/S12649-015-9452-6/FIGURES/6

Bamelis, L., Blancke, S., Camargo-Valero, M., de Clercq, L., Haumont, A., de Keulenaere, B., Delvigne, F., Meers, E., Michels, E., & Ramirez-Sosa, D. (2015). *Techniques for nutrient recovery from digestate derivatives*.

Banks, C. J., Salter, A. M., Heaven, S., & Riley, K. (2011). Energetic and environmental benefits of co-digestion of food waste and cattle slurry: A preliminary assessment. *Resources, Conservation and Recycling*, *56*(1), 71–79. https://doi.org/10.1016/J.RESCONREC.2011.09.006

Barampouti, E. M., Mai, S., Malamis, D., Moustakas, K., & Loizidou, M. (2019). Liquid biofuels from the organic fraction of municipal solid waste: A review. *Renewable and Sustainable Energy Reviews*, *110*, 298–314. https://doi.org/10.1016/J.RSER.2019.04.005

Barampouti, E. M., Mai, S., Malamis, D., Moustakas, K., & Loizidou, M. (2020). Exploring technological alternatives of nutrient recovery from digestate as a secondary resource. *Renewable and Sustainable Energy Reviews*, *134*, 110379. https://doi.org/10.1016/J.RSER.2020.110379

Bauermeister, U., Wild, A., & Meier, T. (2009). Stickstoffabtrennung mit dem ANAStrip-Verfahren System GNS. *Fachagentur Nachwachsende Rohstoffe EV (FNR)(Ed) Gärrestaufbereitung Für Eine Pflanzenbauliche Nutzung-Stand Und F+ E-Bedarf*, *30*, 78–95.

Beal, C. M., Gerber, L. N., Sills, D. L., Huntley, M. E., Machesky, S. C., Walsh, M. J., Tester, J. W., Archibald, I., Granados, J., & Greene, C. H. (2015). Algal biofuel production for fuels and feed in a 100-ha facility: A comprehensive techno-economic analysis and life cycle assessment. *Algal Research*, *10*, 266–279. https://doi.org/10.1016/J.ALGAL.2015.04.017

Bjornsson, W. J., Nicol, R. W., Dickinson, K. E., & McGinn, P. J. (2013). Anaerobic digestates are useful nutrient sources for microalgae cultivation: functional coupling of energy and biomass production. *Journal of Applied Phycology*, *25*(5), 1523–1528.

Boldrin, A., Neidel, T. L., Damgaard, A., Bhander, G. S., Møller, J., & Christensen, T. H. (2011). Modelling of environmental impacts from biological treatment of organic municipal waste in EASEWASTE. *Waste Management*, *31*(4), 619–630.

Bolong, N., Ismail, A. F., Salim, M. R., & Matsuura, T. (2009). A review of the effects of emerging contaminants in wastewater and options for their removal. *Desalination*, *239*(1–3), 229–246. https://doi.org/10.1016/J.DESAL.2008.03.020

Bolzonella, D., Pavan, P., Battistoni, P., & Cecchi, F. (2005). Mesophilic anaerobic digestion of waste activated sludge: influence of the solid retention time in the wastewater treatment process. *Process Biochemistry*, *40*(3–4), 1453–1460. https://doi.org/10.1016/J.PROCBIO.2004.06.036

Bradley, T., Ling-Chin, J., Maga, D., Speranza, L. G., & Roskilly, A. P. (2022). Life Cycle Assessment (LCA) of Algae Biofuels. *Comprehensive Renewable Energy*, 387–404. https://doi.org/10.1016/B978-0-12-819727-1.00067-4

Bullock, C. M., Bicho, P. A., Zhang, Y., & Saddler, J. N. (1996). A solid chemical oxygen demand (COD) method for determining biomass in waste waters. *Water Research*, *30*(5), 1280–1284. https://doi.org/10.1016/0043-1354(95)00271-5

Bumham, J., & Logan, T. (1993). N-VIRO SOIL: ADVANCED ALKALINE SLUDGE STABILIZATION! *Proceedings America Society of Mining and Reclamation*, 765–783.

Cai, T., Park, S. Y., & Li, Y. (2013). Nutrient recovery from wastewater streams by microalgae: status and prospects. *Renewable and Sustainable Energy Reviews*, *19*, 360–369.

Campos, J. C., Moura, D., Costa, A. P., Yokoyama, L., Araujo, F. V. D. F., Cammarota, M. C., & Cardillo, L. (2013). Evaluation of pH, alkalinity and temperature during air stripping process for ammonia removal from landfill leachate. *Http://Dx.Doi.Org/10.1080/10934529.2013.774658*, *48*(9), 1105–1113. https://doi.org/10.1080/10934529.2013.774658

Canada Energy Regulator. (2022). *Provincial and Territorial Energy Profiles*. https://www.cer-rec.gc.ca/en/data-analysis/energy-markets/provincial-territorial-energy-profiles/index.html

Cartes, J., Neumann, P., Hospido, A., & Vidal, G. (2018). Life cycle assessment of management alternatives for sludge from sewage treatment plants in Chile: does advanced anaerobic digestion improve environmental performance compared to current practices? *Journal of Material Cycles and Waste Management*, *20*(3), 1530–1540. https://doi.org/10.1007/S10163-018-0714-9/TABLES/3

Cavinato, C., Bolzonella, D., Pavan, P., Fatone, F., & Cecchi, F. (2013). Mesophilic and thermophilic anaerobic co-digestion of waste activated sludge and source sorted biowaste in pilot- and full-scale reactors. *Renewable Energy*, *55*, 260–265. https://doi.org/10.1016/J.RENENE.2012.12.044

Cecchi, F., Traverso, P., Pavan, P., Bolzonella, D., & Innocenti, L. (2003). Characteristics of the OFMSW and behaviour of the anaerobic digestion process. *Biomethanization of the Organic Fraction of Municipal Solid Wastes*, 141–179.

Cerda, A., Artola, A., Font, X., Barrena, R., Gea, T., & Sánchez, A. (2018). Composting of food wastes: Status and challenges. *Bioresource Technology*, *248*, 57–67. https://doi.org/10.1016/J.BIORTECH.2017.06.133

Cesaro, A., Russo, L., Farina, A., & Belgiorno, V. (2016). Organic fraction of municipal solid waste from mechanical selection: biological stabilization and recovery options. *Environmental Science and Pollution Research*, *23*(2), 1565–1575.

Chaunsali, P., & Peethamparan, S. (2013). Influence of the composition of cement kiln dust on its interaction with fly ash and slag. *Cement and Concrete Research*, *54*, 106–113.

Chen, X., Yan, W., Sheng, K., & Sanati, M. (2014). Comparison of high-solids to liquid anaerobic co-digestion of food waste and green waste. *Bioresource Technology*, *154*, 215–221. https://doi.org/10.1016/J.BIORTECH.2013.12.054

Chinnasamy, S., Bhatnagar, A., Hunt, R. W., & Das, K. C. (2010). Microalgae cultivation in a wastewater dominated by carpet mill effluents for biofuel applications. *Bioresource Technology*, *101*(9), 3097–3105.

Christensen, T. (2011). *Solid waste technology and management*. John Wiley & Sons.

Chuka-ogwude, D., Ogbonna, J., & Moheimani, N. R. (2020). A review on microalgal culture to treat anaerobic digestate food waste effluent. *Algal Research*, *47*, 101841. https://doi.org/10.1016/J.ALGAL.2020.101841

Clarens, A. F., Resurreccion, E. P., White, M. A., & Colosi, L. M. (2010). Environmental life cycle comparison of algae to other bioenergy feedstocks. *Environmental Science and Technology*, *44*(5), 1813–1819. https://doi.org/10.1021/ES902838N/SUPPL_FILE/ES902838N_SI_001.PDF

Cohen, Y., & Kirchmann, H. (2004). Increasing the pH of wastewater to high levels with different gases—CO 2 stripping. *Water, Air, and Soil Pollution*, *159*(1), 265–275.

Colón, J., Martínez-Blanco, J., Gabarell, X., Rieradevall, J., Font, X., Artola, A., & Sánchez, A. (2009). Performance of an industrial biofilter from a composting plant in the removal of ammonia and VOCs after material replacement. *Journal of Chemical Technology & Biotechnology*, *84*(8), 1111–1117. https://doi.org/10.1002/JCTB.2139

Corona, F., Hidalgo, D., Martín-Marroquín, J. M., & Antolín, G. (2020). Study of the influence of the reaction parameters on nutrients recovering from digestate by struvite crystallisation. *Environmental Science and Pollution Research*, 1–13.

Costantini, M., Bacenetti, J., Coppola, G., Orsi, L., Ganzaroli, A., & Guarino, M. (2020). Improvement of human health and environmental costs in the European Union by air scrubbers in intensive pig farming. *Journal of Cleaner Production*, *275*, 124007. https://doi.org/10.1016/J.JCLEPRO.2020.124007

Crutchik, D., & Garrido, J. M. (2011). Struvite crystallization versus amorphous magnesium and calcium phosphate precipitation during the treatment of a saline industrial wastewater. *Water Science and Technology*, *64*(12), 2460–2467.

CWEC. (2019). *Engineering Climate Datasets - Climate - Environment and Climate Change Canada*. Ennvironment Canada . https://climate.weather.gc.ca/prods_servs/engineering_e.html

Davidsson, Å., Gruvberger, C., Christensen, T. H., Hansen, T. L., Jansen, J. la C., & la Cour Jansen, J. (2007). *Methane yield in source-sorted organic fraction of municipal solid waste*. *27*(3), 406–414. https://doi.org/10.1016/J.WASMAN.2006.02.013

de Gioannis, G., & Muntoni, A. (2007). Dynamic transformations of nitrogen during mechanical–biological pre-treatment of municipal solid waste. *Waste Management*, *27*(11), 1479–1485.

de Gioannis, G., Muntoni, A., Cappai, G., & Milia, S. (2009). Landfill gas generation after mechanical biological treatment of municipal solid waste. Estimation of gas generation rate constants. *Waste Management*, *29*(3), 1026–1034.

de Lemos Chernicharo, C. A. (2007). *Anaerobic reactors*. IWA publishing.

de Vries, J. W., & Melse, R. W. (2017). Comparing environmental impact of air scrubbers for ammonia abatement at pig houses: A life cycle assessment. *Biosystems Engineering*, *161*, 53–61. https://doi.org/10.1016/J.BIOSYSTEMSENG.2017.06.010

Debertoldi, M., Vallinni, G., & Pera, A. (1983). The biology of composting: A review. *Waste Management & Research*, *1*(2), 157–176. https://doi.org/10.1016/0734-242X(83)90055-1

Desmidt, E., Ghyselbrecht, K., Zhang, Y., Pinoy, L., der Bruggen, B., Verstraete, W., Rabaey, K., & Meesschaert, B. (2015). Global phosphorus scarcity and full-scale P-recovery techniques: a review. *Critical Reviews in Environmental Science and Technology*, *45*(4), 336–384.

Dhar, B. R., Nakhla, G., & Ray, M. B. (2012). Techno-economic evaluation of ultrasound and thermal pretreatments for enhanced anaerobic digestion of municipal waste activated sludge. *Waste Management*, *32*(3), 542–549. https://doi.org/10.1016/J.WASMAN.2011.10.007

di Maria, F., Micale, C., & Contini, S. (2016). Energetic and environmental sustainability of the co-digestion of sludge with bio-waste in a life cycle perspective. *Applied Energy*, *171*, 67–76. https://doi.org/10.1016/J.APENERGY.2016.03.036

Divert NS. (2018). *Waste Audit Report 2017*. https://divertns.ca/assets/files/WasteAudit2017.pdf

Doublet, J., Francou, C., Poitrenaud, M., & Houot, S. (2011). Influence of bulking agents on organic matter evolution during sewage sludge composting; consequences on compost organic matter stability and N availability. *Bioresource Technology*, *102*(2), 1298–1307.

Drosg, B., Fuchs, W., al Seadi, T., Madsen, M., & Linke, B. (2015). *Nutrient recovery by biogas digestate processing* (Vol. 2015). IEA Bioenergy Dublin.

Du, M., Peng, C., Wang, X., Chen, H., Wang, M., & Zhu, Q. (2017). Quantification of methane emissions from municipal solid waste landfills in China during the past decade. *Renewable and Sustainable Energy Reviews*, *78*, 272–279.

Duan, N., Dong, B., Wu, B., & Dai, X. (2012a). High-solid anaerobic digestion of sewage sludge under mesophilic conditions: Feasibility study. *Bioresource Technology*, *104*, 150–156. https://doi.org/10.1016/J.BIORTECH.2011.10.090

Duan, N., Dong, B., Wu, B., & Dai, X. (2012b). High-solid anaerobic digestion of sewage sludge under mesophilic conditions: Feasibility study. *Bioresource Technology*, *104*, 150–156. https://doi.org/10.1016/J.BIORTECH.2011.10.090

Duan, N., Khoshnevisan, B., Lin, C., Liu, Z., & Liu, H. (2020). Life cycle assessment of anaerobic digestion of pig manure coupled with different digestate treatment technologies. *Environment International*, *137*, 105522. https://doi.org/10.1016/J.ENVINT.2020.105522

Dumont, É. (2018). Impact of the treatment of NH3 emissions from pig farms on greenhouse gas emissions. Quantitative assessment from the literature data. *New Biotechnology*, *46*, 31–37. https://doi.org/10.1016/J.NBT.2018.06.001

Edwards, J., Othman, M., Crossin, E., & Burn, S. (2017). Anaerobic co-digestion of municipal food waste and sewage sludge: A comparative life cycle assessment in the context of a waste service provision. *Bioresource Technology*, *223*, 237–249. https://doi.org/10.1016/J.BIORTECH.2016.10.044

Egle, L., Rechberger, H., & Zessner, M. (2015). Overview and description of technologies for recovering phosphorus from municipal wastewater. *Resources, Conservation and Recycling*, *105*, 325–346.

Eisa, H. M., Vaezi, I., & Ardakani, A. M. (2019). Evaluation of solidification/stabilization in arsenic-contaminated soils using lime dust and cement kiln dust. *Bulletin of Engineering Geology and the Environment*, 1–10.

El-Fadel, M., Findikakis, A. N., & Leckie, J. O. (1997). *Environmental impacts of solid waste landfilling*. *50*(1), 1–25. https://doi.org/10.1006/JEMA.1995.0131

Enamala, M. K., Enamala, S., Chavali, M., Donepudi, J., Yadavalli, R., Kolapalli, B., Aradhyula, T. V., Velpuri, J., & Kuppam, C. (2018). Production of biofuels from microalgae-A review on cultivation, harvesting, lipid extraction, and numerous applications of microalgae. *Renewable and Sustainable Energy Reviews*, *94*, 49–68.

Environment Canada. (2018). *Canadian Environmental Sustainability Indicators: Solid waste diversion and disposal*. www.canada.ca/en/environment-climate-change/services/environmental-

Epstein, E. (2011). Industrial composting: Environmental engineering and facilities management. In *Environmental engineering and facilities management. New York: Taylor and Francis Group*. Taylor and Francis Group.

Errico, M., Fjerbaek Sotoft, L., Kjærhuus Nielsen, A., & Norddahl, B. (2018). Treatment costs of ammonia recovery from biogas digestate by air stripping analyzed by process simulation. *Clean Technologies and Environmental Policy*, *20*(7), 1479–1489. https://doi.org/10.1007/S10098-017-1468-0/TABLES/5

EU Commission. (2010). International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. In *European Commission - Joint Research Centre - Institute for Environment and Sustainability*. Publications Office of the European Union. https://eplca.jrc.ec.europa.eu/uploads/ILCD-Handbook-General-guide-for-LCA-DETAILED-GUIDANCE-12March2010-ISBN-fin-v1.0-EN.pdf

Fatta, D., & Mol, S. (2003). *Assessment of information related to waste and material flows - a catalogue of methods and tools — European Environment Agency*. https://www.eea.europa.eu/publications/technical_report_2003_96

Fernández-Nava, Y., del Río, J., Rodríguez-Iglesias, J., Castrillón, L., & Marañón, E. (2014). Life cycle assessment of different municipal solid waste management options: a case study of Asturias (Spain). *Journal of Cleaner Production*, *81*, 178–189. https://doi.org/10.1016/J.JCLEPRO.2014.06.008

Finnveden, G. (1999). Methodological aspects of life cycle assessment of integrated solid waste management systems. *Resources, Conservation and Recycling*, *26*(3–4), 173–187. https://doi.org/10.1016/S0921-3449(99)00005-1

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., & Suh, S. (2009). Recent developments in Life Cycle Assessment. *Journal of Environmental Management*, *91*(1), 1–21. https://doi.org/10.1016/J.JENVMAN.2009.06.018

Fisher, R. M., Alvarez-Gaitan, J. P., & Stuetz, R. M. (2019). Review of the effects of wastewater biosolids stabilization processes on odor emissions. *Critical Reviews in Environmental Science and Technology*, *49*(17), 1515–1586.

Forster-Carneiro, T., Pérez, M., & Romero, L. I. (2008). Anaerobic digestion of municipal solid wastes: Dry thermophilic performance. *Bioresource Technology*, *99*(17), 8180–8184. https://doi.org/10.1016/J.BIORTECH.2008.03.021

Foteinis, S., Antoniadis-Gavriil, A., & Tsoutsos, T. (2018). Life cycle assessment of algae-to-biodiesel shallow pond production systems in the Mediterranean: influence of species, pond type, by(co)-product valorisation and electricity mix. *Biofuels, Bioproducts and Biorefining*, *12*(4), 542–558. https://doi.org/10.1002/BBB.1871

Fuchs, W., & Drosg, B. (2013). Assessment of the state of the art of technologies for the processing of digestate residue from anaerobic digesters. *Water Science and Technology*, *67*(9), 1984–1993.

Gartner Lee Ltd. (2007). *Study of Stabilized Landfill Final Report*. http://www.durhamenvironmentwatch.org/Incinerator%20Files/FinalStabilizedLandfillGartnerLee.pdf

Gienau, T., Brüß, U., Kraume, M., & Rosenberger, S. (2018). Nutrient recovery from biogas digestate by optimised membrane treatment. *Waste and Biomass Valorization*, *9*(12), 2337–2347.

Giroux, L. (2014). *State of Waste Management in Canada*. Canadian Council of Ministers of the Environment.

https://www.ccme.ca/files/Resources/waste/wst_mgmt/State_Waste_Mgmt_in_Canada%20April%202015%20revised.pdf

GIROVIČ, M. J. (1996). *Biosolids Treatment and Management: Processes for Beneficial Use*. CRC Press.

Gómez, X., Cuetos, M. J., Cara, J., Morán, A., & García, A. I. (2006). Anaerobic co-digestion of primary sludge and the fruit and vegetable fraction of the municipal solid wastes: Conditions for mixing and evaluation of the organic loading rate. *Renewable Energy*, *31*(12), 2017–2024. https://doi.org/10.1016/J.RENENE.2005.09.029

Gottardo, M., Micolucci, F., Bolzonella, D., Uellendahl, H., & Pavan, P. (2017). Pilot scale fermentation coupled with anaerobic digestion of food waste - Effect of dynamic digestate recirculation. *Renewable Energy*, *114*, 455–463. https://doi.org/10.1016/J.RENENE.2017.07.047

GREET. (2011). *Life-cycle analysis of algal lipid fuels with the greet model*.

Guilayn, F., Jimenez, J., Rouez, M., Crest, M., & Patureau, D. (2019). Digestate mechanical separation: Efficiency profiles based on anaerobic digestion feedstock and equipment choice. *Bioresource Technology*, *274*, 180–189. https://doi.org/10.1016/J.BIORTECH.2018.11.090

Guilayn, F., Rouez, M., Crest, M., Patureau, D., & Jimenez, J. (2020). Valorization of digestates from urban or centralized biogas plants: a critical review. *Reviews in Environmental Science and Bio/Technology*.

Guštin, S., & Marinšek-Logar, R. (2011). Effect of pH, temperature and air flow rate on the continuous ammonia stripping of the anaerobic digestion effluent. *Process Safety and Environmental Protection*, *89*(1), 61–66. https://doi.org/10.1016/J.PSEP.2010.11.001

Guven, H., Wang, Z., & Eriksson, O. (2019). Evaluation of future food waste management alternatives in Istanbul from the life cycle assessment perspective. *Journal of Cleaner Production*, *239*, 117999. https://doi.org/10.1016/J.JCLEPRO.2019.117999

Halifax Water. (2020a). *Personal Communications, Mill Cove wastewater treatment facility data* .

Halifax Water. (2020b). *Provided data on biosolids processing facility (personal communication)*.

Halifax Water. (2020c). Halifax Water 2020/21 Business Plan. *Halifax Water, Halifax, NS*. https://www.halifax.ca/sites/default/files/documents/city-hall/regional-council/200225rc1332.pdf

Hallas, J. F., Mackowiak, C. L., Wilkie, A. C., & Harris, W. G. (2019). Struvite Phosphorus Recovery from Aerobically Digested Municipal Wastewater. *Sustainability 2019, Vol. 11, Page 376*, *11*(2), 376. https://doi.org/10.3390/SU11020376

Hamoda, M. F., Abu Qdais, H. A., & Newham, J. (1998). Evaluation of municipal solid waste composting kinetics. *Resources, Conservation and Recycling*, *23*(4), 209–223. https://doi.org/10.1016/S0921-3449(98)00021-4

Hansen, T. L., Bhander, G. S., Christensen, T. H., Bruun, S., & Jensen, L. S. (2006). Life cycle modelling of environmental impacts of application of processed organic municipal solid waste on agricultural land (EASEWASTE). *Waste Management & Research*, *24*(2), 153–166.

Hansen, T. L., la Cour Jansen, J., Spliid, H., Davidsson, Å., & Christensen, T. H. (2007). Composition of source-sorted municipal organic waste collected in Danish cities. *Waste Management*, *27*(4), 510–518.

Havukainen, J., Saud, A., Astrup, T. F., Peltola, P., & Horttanainen, M. (2022a). Environmental performance of dewatered sewage sludge digestate utilization based on life cycle assessment. *Waste Management*, *137*, 210–221. https://doi.org/10.1016/J.WASMAN.2021.11.005

Havukainen, J., Saud, A., Astrup, T. F., Peltola, P., & Horttanainen, M. (2022b). Environmental performance of dewatered sewage sludge digestate utilization based on life cycle assessment. *Waste Management*, *137*, 210–221. https://doi.org/10.1016/J.WASMAN.2021.11.005

Herrera, A., D'Imporzano, G., Zilio, M., Pigoli, A., Rizzi, B., Meers, E., Schouman, O., Schepis, M., Barone, F., Giordano, A., & Adani, F. (2022). Environmental Performance in the Production and Use of Recovered Fertilizers from Organic Wastes Treated by Anaerobic Digestion vs Synthetic Mineral Fertilizers. *ACS Sustainable Chemistry and Engineering*, *10*(2), 986–997. https://doi.org/10.1021/ACSSUSCHEMENG.1C07028/ASSET/IMAGES/LARGE/SC1C07028_0005.JPEG

HRM. (2011). *Municipal Usage of N-Viro Halifax Soil Amendment*. Halifax Regional Council, Nova Scotia, Canada. http://legacycontent.halifax.ca/council/agendasc/documents/111025ca1112.pdf

HRM. (2016). *Organics Processing and Management*. https://www.halifax.ca/sites/default/files/documents/city-hall/regional-council/160223rc1431.pdf

HRM. (2020a). *Green carts, leaf and yard material*. https://www.halifax.ca/home-property/garbage-recycling-green-cart/green-carts-leaf-yard-material

HRM. (2020b). *Halifax Solid Waste*.

Hu, Y., Shen, F., Yuan, H., Zou, D., Pang, Y., Liu, Y., Zhu, B., Chufo, W. A., Jaffar, M., & Li, X. (2014). Influence of recirculation of liquid fraction of the digestate (LFD) on maize stover anaerobic digestion. *Biosystems Engineering*, *127*, 189–196. https://doi.org/10.1016/J.BIOSYSTEMSENG.2014.09.006

Huerta-Pujol, O., Gallart, M., Soliva, M., Martínez-Farré, F. X., & López, M. (2011). Effect of collection system on mineral content of biowaste. *Resources, Conservation and Recycling*, *55*(11), 1095–1099. https://doi.org/10.1016/J.RESCONREC.2011.06.008

Hydromantis Inc. (2010). *EMERGING SUBSTANCES OF CONCERN IN BIOSOLIDS: CONCENTRATIONS AND EFFECTS OF TREATMENT PROCESSES*. CANADIAN COUNCIL OF MINISTERS OF THE ENVIRONMENT. https://www.ccme.ca/files/Resources/waste/biosolids/pn_1440_contam_invt_rvw.pdf

Igoni, A. H., Ayotamuno, M. J., Eze, C. L., Ogaji, S. O. T., & Probert, S. D. (2008). Designs of anaerobic digesters for producing biogas from municipal solid-waste. *Applied Energy*, *85*(6), 430–438.

ISO. (2006). *ISO - ISO 14040:2006 - Environmental management — Life cycle assessment — Principles and framework*. https://www.iso.org/standard/37456.html

Iyovo, G. D., Du, G., & Chen, J. (2010). Sustainable bioenergy bioprocessing: biomethane production, digestate as biofertilizer and as supplemental feed in algae cultivation to promote algae biofuel commercialization. *J. Microb. Biochem. Technol*, *2*(4), 100–106.

Jamaludin, Z., Rollings-Scattergood, S., Lutes, K., & Vaneeckhaute, C. (2018). Evaluation of sustainable scrubbing agents for ammonia recovery from anaerobic digestate. *Bioresource Technology*, *270*, 596–602. https://doi.org/10.1016/J.BIORTECH.2018.09.007

Jensen, M. B., Møller, J., & Scheutz, C. (2016). Comparison of the organic waste management systems in the Danish–German border region using life cycle assessment (LCA). *Waste Management*, *49*, 491–504.

Jiang, Y., McAdam, E., Zhang, Y., Heaven, S., Banks, C., & Longhurst, P. (2019). Ammonia inhibition and toxicity in anaerobic digestion: A critical review. *Journal of Water Process Engineering*, *32*, 100899. https://doi.org/10.1016/J.JWPE.2019.100899

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Schleiss, K., & Spielmann, M. (2007). Life cycle inventories of bioenergy. ecoinvent report No. 17. In *Final report ecoinvent data v2. 0* (Vol. 17).

Karmperis, A. C., Aravossis, K., Tatsiopoulos, I. P., & Sotirchos, A. (2013). Decision support models for solid waste management: Review and game-theoretic approaches. *Waste Management*, *33*(5), 1290–1301. https://doi.org/10.1016/J.WASMAN.2013.01.017

Kataki, S., West, H., Clarke, M., & Baruah, D. C. (2016). Phosphorus recovery as struvite from farm, municipal and industrial waste: Feedstock suitability, methods and pre-treatments. *Waste Management*, *49*, 437–454.

Kaza, S., Yao, L., Bhada-Tata, P., & van Woerden, F. (2018). *What a waste 2.0: a global snapshot of solid waste management to 2050*. The World Bank.

Khalid, A. A. H., Yaakob, Z., Abdullah, S. R. S., & Takriff, M. S. (2018). Growth improvement and metabolic profiling of native and commercial Chlorella sorokiniana strains acclimatized in recycled agricultural wastewater. *Bioresource Technology*, *247*, 930–939. https://doi.org/10.1016/J.BIORTECH.2017.09.195

Khan, A. H., López-Maldonado, E. A., Khan, N. A., Villarreal-Gómez, L. J., Munshi, F. M., Alsabhan, A. H., & Perveen, K. (2022). Current solid waste management strategies and energy recovery in developing countries - State of art review. *Chemosphere*, *291*, 133088. https://doi.org/10.1016/J.CHEMOSPHERE.2021.133088

Khanal, S. K. (2011). *Anaerobic biotechnology for bioenergy production: principles and applications*. John Wiley & Sons.

Kim, D.-H., & Oh, S.-E. (2011a). Continuous high-solids anaerobic co-digestion of organic solid wastes under mesophilic conditions. *Waste Management*, *31*(9–10), 1943–1948.

Kim, D.-H., & Oh, S.-E. (2011b). Continuous high-solids anaerobic co-digestion of organic solid wastes under mesophilic conditions. *Waste Management*, *31*(9–10), 1943–1948.

Kim, S., Lee, Y., & Hwang, S. J. (2013). Removal of nitrogen and phosphorus by Chlorella sorokiniana cultured heterotrophically in ammonia and nitrate. *International Biodeterioration & Biodegradation*, *85*, 511–516. https://doi.org/10.1016/J.IBIOD.2013.05.025

Kobayashi, N., Noel, E. A., Barnes, A., Watson, A., Rosenberg, J. N., Erickson, G., & Oyler, G. A. (2013). Characterization of three Chlorella sorokiniana strains in anaerobic digested effluent from cattle manure. *Bioresource Technology*, *150*, 377–386. https://doi.org/10.1016/J.BIORTECH.2013.10.032

Komilis, D. P., Ham, R. K., & Stegmann, R. (1999). The effect of municipal solid waste pretreatment on landfill behavior: a literature review. *Waste Management and Research*, *17*(1), 10–19.

Kumar, A., & Samadder, S. R. (2020). Performance evaluation of anaerobic digestion technology for energy recovery from organic fraction of municipal solid waste: A review. *Energy*, *197*, 117253. https://doi.org/10.1016/J.ENERGY.2020.117253

Kumar, S., Gaikwad, S. A., Shekdar, A. v, Kshirsagar, P. S., & Singh, R. N. (2004). Estimation method for national methane emission from solid waste landfills. *Atmospheric Environment*, *38*(21), 3481–3487.

la Cour Jansen, J., Spliid, H., Hansen, T. L., Svärd, Å., & Christensen, T. H. (2004). Assessment of sampling and chemical analysis of source-separated organic household waste. *Waste Management*, *24*(6), 541–549. https://doi.org/10.1016/J.WASMAN.2004.02.013

Lamolinara, B., Pérez-Martínez, A., Guardado-Yordi, E., Guillén Fiallos, C., Diéguez-Santana, K., & Ruiz-Mercado, G. J. (2022). Anaerobic digestate management, environmental impacts, and techno-economic challenges. *Waste Management*, *140*, 14–30. https://doi.org/10.1016/J.WASMAN.2021.12.035

Lei, X., Sugiura, N., Feng, C., & Maekawa, T. (2007). Pretreatment of anaerobic digestion effluent with ammonia stripping and biogas purification. *Journal of Hazardous Materials*, *145*(3), 391–397. https://doi.org/10.1016/J.JHAZMAT.2006.11.027

Leite, L. de S., Hoffmann, M. T., & Daniel, L. A. (2019). Microalgae cultivation for municipal and piggery wastewater treatment in Brazil. *Journal of Water Process Engineering*, *31*, 100821. https://doi.org/10.1016/J.JWPE.2019.100821

Li, H., Jin, C., Zhang, Z., O'Hara, I., & Mundree, S. (2017). Environmental and economic life cycle assessment of energy recovery from sewage sludge through different anaerobic digestion pathways. *Energy*, *126*, 649–657. https://doi.org/10.1016/J.ENERGY.2017.03.068

Li, H., Si, D., Liu, C., Feng, K., & Liu, C. (2018). Performance of direct anaerobic digestion of dewatered sludge in long-term operation. *Bioresource Technology*, *250*, 355–364. https://doi.org/10.1016/J.BIORTECH.2017.11.075

Li, Y., Manandhar, A., Li, G., & Shah, A. (2018). Life cycle assessment of integrated solid state anaerobic digestion and composting for on-farm organic residues treatment. *Waste Management*, *76*, 294–305. https://doi.org/10.1016/J.WASMAN.2018.03.025

Lin, Z. (2020). *Evaluation of Seasonal Dynamics of Soil Macro-Nutrients and Corn Nutrient Uptake in Fields Amended with Three Types of Municipal Biosolids*. Dalhousie University.

Liu, B., & Rajagopal, D. (2019). Life-cycle energy and climate benefits of energy recovery from wastes and biomass residues in the United States. *Nature Energy 2019 4:8*, *4*(8), 700–708. https://doi.org/10.1038/s41560-019-0430-2

Liu, C., Wang, W., Anwar, N., Ma, Z., Liu, G., & Zhang, R. (2017). Effect of organic loading rate on anaerobic digestion of food waste under mesophilic and thermophilic conditions. *Energy & Fuels*, *31*(3), 2976–2984.

Liu, X., Li, R., & Ji, M. (2019). Effects of Two-Stage Operation on Stability and Efficiency in Co-Digestion of Food Waste and Waste Activated Sludge. *Energies 2019, Vol. 12, Page 2748*, *12*(14), 2748. https://doi.org/10.3390/EN12142748

Liu, X., Wen, G., Hu, Z., & Wang, J. (2018). Coupling effects of pH and Mg/P ratio on P recovery from anaerobic digester supernatant by struvite formation. *Journal of Cleaner Production*, *198*, 633–641. https://doi.org/10.1016/J.JCLEPRO.2018.07.073

Lizzul, A. M., Hellier, P., Purton, S., Baganz, F., Ladommatos, N., & Campos, L. (2014). Combined remediation and lipid production using Chlorella sorokiniana grown on wastewater and exhaust gases. *Bioresource Technology*, *151*, 12–18. https://doi.org/10.1016/J.BIORTECH.2013.10.040

Logan, T. J., & Harrison, B. J. (1995). Physical Characteristics of Alkaline Stabilized Sewage Sludge (N-Viro Soil) and Their Effects on Soil Physical Properties. *Journal of Environmental Quality*, *24*(1), 153–164. https://doi.org/10.2134/JEQ1995.00472425002400010022X

Lorick, D., Macura, B., Ahlström, M., Grimvall, A., & Harder, R. (2020). Effectiveness of struvite precipitation and ammonia stripping for recovery of phosphorus and nitrogen from anaerobic digestate: a systematic review. *Environmental Evidence*, *9*(1), 1–20. https://doi.org/10.1186/S13750-020-00211-X/TABLES/5

LP Consulting Ltd. (2017). *Creating an Agricultural Market for Nova Scotia Compost-Final Report*. https://divertns.ca/assets/files/Creating-an-Agricultural-Market-for-Nova-Scotia-Compost-LP-Consulting-Limited.pdf

Malhotra, M., Aboudi, K., Pisharody, L., Singh, A., Banu, J. R., Bhatia, S. K., Varjani, S., Kumar, S., González-Fernández, C., Kumar, S., Singh, R., & Tyagi, V. K. (2022). Biorefinery of anaerobic digestate in a circular bioeconomy: Opportunities, challenges and perspectives. *Renewable and Sustainable Energy Reviews*, *166*, 112642. https://doi.org/10.1016/J.RSER.2022.112642

Manu, M. K., Li, D., Liwen, L., Jun, Z., Varjani, S., & Wong, J. W. C. (2021). A review on nitrogen dynamics and mitigation strategies of food waste digestate composting. *Bioresource Technology*, *334*, 125032. https://doi.org/10.1016/J.BIORTECH.2021.125032

Mao, C., Feng, Y., Wang, X., & Ren, G. (2015). Review on research achievements of biogas from anaerobic digestion. *Renewable and Sustainable Energy Reviews*, *45*, 540–555. https://doi.org/10.1016/J.RSER.2015.02.032

Maslehuddin, M., Al-Amoudi, O. S. B., Shameem, M., Rehman, M. K., & Ibrahim, M. (2008). Usage of cement kiln dust in cement products–research review and preliminary investigations. *Construction and Building Materials*, *22*(12), 2369–2375.

Maulini-Duran, C., Artola, A., Font, X., & Sánchez, A. (2014). Gaseous emissions in municipal wastes composting: effect of the bulking agent. *Bioresource Technology*, *172*, 260–268.

Méndez-Acosta, H. O., Snell-Castro, R., Alcaraz-González, V., González-Álvarez, V., & Pelayo-Ortiz, C. (2010). Anaerobic treatment of Tequila vinasses in a CSTR-type digester. *Biodegradation*, *21*(3), 357–363. https://doi.org/10.1007/S10532-009-9306-7/FIGURES/2

Metcalf & Eddy, & Tchobanoglous, G. (2014). *Wastewater Engineering: Treatment and Resource Recovery*. McGraw-Hill.

Micolucci, F., Gottardo, M., Pavan, P., Cavinato, C., & Bolzonella, D. (2018). Pilot scale comparison of single and double-stage thermophilic anaerobic digestion of food waste. *Journal of Cleaner Production*, *171*, 1376–1385.

Mirror NS. (2017). *Otter Lake Waste Processing and Disposal Facilities-Site Development and Operations-Annual Report*. http://www.otterlakecmc.ca/wp-content/uploads/2020/05/Annual-Report-2017.pdf

Mirror NS. (2018). *Otter Lake Waste Processing and Disposal Facilities-Site Development and Operations-Annual Report*. http://www.otterlakecmc.ca/wp-content/uploads/2020/05/Annual-Report-2018.pdf

Mirror NS. (2019). *Otter Lake Waste Processing and Disposal Facilities-Site Development and Operations-Annual Report*. http://www.otterlakecmc.ca/wp-content/uploads/2020/05/HRM-Annual-Report-2019.pdf

M.N.V Prasad. (2016). *Bioremediation and Bioeconomy*. Elsevier. https://www.sciencedirect.com/book/9780128028308/bioremediation-and-bioeconomy#book-description

Mohee, R., & Mudhoo, A. (2005). Analysis of the physical properties of an in-vessel composting matrix. *Powder Technology*, *155*(1), 92–99.

Molina Grima, E., Belarbi, E. H., Acién Fernández, F. G., Robles Medina, A., & Chisti, Y. (2003). Recovery of microalgal biomass and metabolites: process options and economics. *Biotechnology Advances*, *20*(7–8), 491–515. https://doi.org/10.1016/S0734-9750(02)00050-2

Möller, K., & Müller, T. (2012). Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review. *Engineering in Life Sciences*, *12*(3), 242–257. https://doi.org/10.1002/ELSC.201100085

Monlau, F., Sambusiti, C., Ficara, E., Aboulkas, A., Barakat, A., & Carrère, H. (2015). New opportunities for agricultural digestate valorization: current situation and perspectives. *Energy & Environmental Science*, *8*(9), 2600–2621.

Montejo, C., Tonini, D., del Carmen Márquez, M., & Astrup, T. F. (2013). Mechanical–biological treatment: Performance and potentials. An LCA of 8 MBT plants including waste characterization. *Journal of Environmental Management*, *128*, 661–673.

Morales, N., Boehler, M. A., Buettner, S., Liebi, C., & Siegrist, H. (2013). Recovery of N and P from Urine by Struvite Precipitation Followed by Combined Stripping with Digester Sludge Liquid at Full Scale. *Water 2013, Vol. 5, Pages 1262-1278*, *5*(3), 1262–1278. https://doi.org/10.3390/W5031262

Moran, S. (2018). Applied Guide to Water and Effluent Treatment Plant Design. In *Elsevier* . Elsevier.

Morris, J., Scott Matthews, H., & Morawski, C. (2013). Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. *Waste Management*, *33*(3), 545–551. https://doi.org/10.1016/J.WASMAN.2012.08.004

Mukherjee, S., Mukhopadhyay, S., Hashim, M. A., & sen Gupta, B. (2015). Contemporary environmental issues of landfill leachate: assessment and remedies. *Critical Reviews in Environmental Science and Technology*, *45*(5), 472–590.

Nagao, N., Tajima, N., Kawai, M., Niwa, C., Kurosawa, N., Matsuyama, T., Yusoff, F. M., & Toda, T. (2012). Maximum organic loading rate for the single-stage wet anaerobic digestion of food waste. *Bioresource Technology*, *118*, 210–218. https://doi.org/10.1016/J.BIORTECH.2012.05.045

National Institute for Public Health and the Environment. (2017). *ReCiPe 2016: A harmonized life cycle impact assessment method at midpoint and endpoint level,* *Report I: Characterization*. https://pre-sustainability.com/legacy/download/Report_ReCiPe_2017.pdf

Nghiem, L. D., Koch, K., Bolzonella, D., & Drewes, J. E. (2017). Full scale co-digestion of wastewater sludge and food waste: Bottlenecks and possibilities. *Renewable and Sustainable Energy Reviews*, *72*, 354–362. https://doi.org/10.1016/J.RSER.2017.01.062

Nguyen, M.-L. T., Lin, C.-Y., & Lay, C.-H. (2019). Microalgae cultivation using biogas and digestate carbon sources. *Biomass and Bioenergy*, *122*, 426–432.

Ni, P., Lyu, T., Sun, H., Dong, R., & Wu, S. (2017). Liquid digestate recycled utilization in anaerobic digestion of pig manure: Effect on methane production, system stability and heavy metal mobilization. *Energy*, *141*, 1695–1704. https://doi.org/10.1016/J.ENERGY.2017.11.107

Nieminen, J. (2010). Phosphorus recovery and recycling from municipal wastewater sludge. *Vesitalous*, *3*, 15–17.

Noor, Z. Z., Yusuf, R. O., Abba, A. H., Abu Hassan, M. A., & Mohd Din, M. F. (2013). An overview for energy recovery from municipal solid wastes (MSW) in Malaysia scenario. *Renewable and Sustainable Energy Reviews*, *20*, 378–384. https://doi.org/10.1016/J.RSER.2012.11.050

Nova Scotia Environment. (1994). *Environment Act C1*. https://nslegislature.ca/sites/default/files/legc/statutes/environment.pdf

Nova Scotia Environment. (2007). *Environmental Goals and Sustainable Prosperity Act c. 7*.

NS Enviro. (2009). *Renewal of Nova Scotia's Solid Waste Resource Management Strategy, Consultation Summary Report*. Nova Scotia Environment, Nova Scotia, Canada. https://novascotia.ca/nse/waste/docs/SolidWasteStrategy.2009.Renewal.pdf

NS Government. (2015). *Our Electricity Future Nova Scotia's Electricity Plan 2015-2040*.

NS Government. (2020a). *Compost Management Tonnage and Cost*. https://data.novascotia.ca/Environment-and-Energy/Compost-Management-Tonnage-and-Cost/4egx-2bpq/data

NS Government. (2020b). *Municipal Solid Waste Curbside Collection Tonnage*. https://data.novascotia.ca/Environment-and-Energy/Municipal-Solid-Waste-Curbside-Collection-Tonnage/r7ci-8szi

NS Government. (2020c). *Municipal Solid Waste Disposal Totals*. Municipal

NS Government. (2020d). *Organic Composting Facilities in Nova Scotia*. https://novascotia.ca/nse/waste.facilities/facilities.organic.composting.asp

NSE. (2016). *Operations of the Halifax Ragged Lake Source Separated Organics Compost Facility (Facility)*. https://www.halifax.ca/sites/default/files/documents/city-hall/boards-committees-commissions/170927wcac911ii.pdf

N-Viro Systems Canada Inc. (2007). *Innovative Sewage Treatment And Resource Recovery Technology For Victoria, British Columbia, Canada*. https://www.crd.bc.ca/docs/default-source/seaterra-pdf/related-articles/techreview_sludge-stablilzation—nviro_submission.pdf?sfvrsn=4c6565ca_2

Ostrem, K., & Themelis, N. J. (2004). Greening waste: Anaerobic digestion for treating the organic fraction of municipal solid wastes. *Earth Engineering Center Columbia University*, 6–9.

Palakodeti, A., Azman, S., Rossi, B., Dewil, R., & Appels, L. (2021). A critical review of ammonia recovery from anaerobic digestate of organic wastes via stripping. *Renewable and Sustainable Energy Reviews*, *143*, 110903. https://doi.org/10.1016/J.RSER.2021.110903

Peng, X., Nges, I. A., & Liu, J. (2016). Improving methane production from wheat straw by digestate liquor recirculation in continuous stirred tank processes. *Renewable Energy*, *85*, 12–18. https://doi.org/10.1016/J.RENENE.2015.06.023

Petric, I., & Selimbašić, V. (2008). Development and validation of mathematical model for aerobic composting process. *Chemical Engineering Journal*, *139*(2), 304–317. https://doi.org/10.1016/J.CEJ.2007.08.017

Piaia, B. (2013). Full-scale membrane filtration system for the treatment of digestate from a co-digestion plant. *Applied Engineering in Agriculture*, *29*(6), 985–990.

Połomka, J., & Jędrczak, A. (2019). Efficiency of waste processing in the MBT system. *Waste Management*, *96*, 9–14.

Preston, M. (2022). *Otter Lake compost sorter to be deactivated* . CBC News. https://www.cbc.ca/news/canada/nova-scotia/otter-lake-compost-sorter-to-be-deactivated-1.6394714

Provolo, G., Perazzolo, F., Mattachini, G., Finzi, A., Naldi, E., & Riva, E. (2017). Nitrogen removal from digested slurries using a simplified ammonia stripping technique. *Waste Management*, *69*, 154–161. https://doi.org/10.1016/J.WASMAN.2017.07.047

Rahaman, M. S., Ellis, N., & Mavinic, D. S. (2008). Effects of various process parameters on struvite precipitation kinetics and subsequent determination of rate constants. *Water Science and Technology*, *57*(5), 647–654. https://doi.org/10.2166/WST.2008.022

Richter, A., Bruce, N., Ng, K. T. W., Chowdhury, A., & Vu, H. L. (2017). Comparison between Canadian and Nova Scotian waste management and diversion models—A Canadian case study. *Sustainable Cities and Society*, *30*, 139–149.

Rodríguez, N. H., Granados, R. J., Blanco-Varela, M. T., Cortina, J. L., Martínez-Ramírez, S., Marsal, M., Guillem, M., Puig, J., Fos, C., & Larrotcha, E. (2012). Evaluation of a lime-mediated sewage sludge stabilisation process. Product characterisation and technological validation for its use in the cement industry. *Waste Management*, *32*(3), 550–560.

Rodriguez-Narvaez, O. M., Peralta-Hernandez, J. M., Goonetilleke, A., & Bandala, E. R. (2017). Treatment technologies for emerging contaminants in water: A review. *Chemical Engineering Journal*, *323*, 361–380. https://doi.org/10.1016/J.CEJ.2017.04.106

Roy, M. M., Dutta, A., Corscadden, K., Havard, P., & Dickie, L. (2011). Review of biosolids management options and co-incineration of a biosolid-derived fuel. *Waste Management*, *31*(11), 2228–2235. https://doi.org/10.1016/J.WASMAN.2011.06.008

Sayedin, F., Kermanshahi-pour, A., & He, Q. S. (2019). Evaluating the potential of a novel anaerobic baffled reactor for anaerobic digestion of thin stillage: Effect of organic loading rate, hydraulic retention time and recycle ratio. *Renewable Energy*, *135*, 975–983. http://dx.doi.org/10.1016/j.renene.2018.12.084

Sayedin, F., Kermanshahi-pour, A., He, Q. S., Tibbetts, S. M., Lalonde, C. G. E., & Brar, S. K. (2020). Microalgae cultivation in thin stillage anaerobic digestate for nutrient recovery and bioproduct production. *Algal Research*, *47*, 101867. https://doi.org/10.1016/j.algal.2020.101867

Scaglia, B., Salati, S., di Gregorio, A., Carrera, A., Tambone, F., & Adani, F. (2013). Short mechanical biological treatment of municipal solid waste allows landfill impact reduction saving waste energy content. *Bioresource Technology*, *143*, 131–138.

Schievano, A., D'Imporzano, G., Salati, S., & Adani, F. (2011). On-field study of anaerobic digestion full-scale plants (Part I): An on-field methodology to determine mass, carbon and nutrients balance. *Bioresource Technology*, *102*(17), 7737–7744. https://doi.org/10.1016/J.BIORTECH.2011.06.006

Schrijvers, D. L., Loubet, P., & Sonnemann, G. (2016). Critical review of guidelines against a systematic framework with regard to consistency on allocation procedures for recycling in LCA. *International Journal of Life Cycle Assessment*, *21*(7), 994–1008. https://doi.org/10.1007/S11367-016-1069-X/TABLES/1

Serna-Maza, A., Heaven, S., & Banks, C. J. (2015). Biogas stripping of ammonia from fresh digestate from a food waste digester. *Bioresource Technology*, *190*, 66–75. https://doi.org/10.1016/J.BIORTECH.2015.04.041

Sheets, J. P., Yang, L., Ge, X., Wang, Z., & Li, Y. (2015). Beyond land application: Emerging technologies for the treatment and reuse of anaerobically digested agricultural and food waste. *Waste Management*, *44*, 94–115.

Siddique, M. N. I., & Wahid, Z. A. (2018). Achievements and perspectives of anaerobic co-digestion: A review. *Journal of Cleaner Production*, *194*, 359–371. https://doi.org/10.1016/J.JCLEPRO.2018.05.155

Singh, M., Reynolds, D. L., & Das, K. C. (2011). Microalgal system for treatment of effluent from poultry litter anaerobic digestion. *Bioresource Technology*, *102*(23), 10841–10848. https://doi.org/10.1016/J.BIORTECH.2011.09.037

Slorach, P. C., Jeswani, H. K., Cuéllar-Franca, R., & Azapagic, A. (2019). Environmental sustainability of anaerobic digestion of household food waste.

Journal of Environmental Management, *236*, 798–814. https://doi.org/10.1016/J.JENVMAN.2019.02.001

SNC-Lavalin Inc. (2013). *A Peer Review of the January 2013 Stantec Report "Waste Resource Strategy Update."*

Stantec. (2013). *Waste Resource Strategy Update - Halifax Regional Municipality*. Stantec Consulting Co. https://www.shapeyourcityhalifax.ca/301/widgets/2403/documents/460

Stiles, W. A. v, Styles, D., Chapman, S. P., Esteves, S., Bywater, A., Melville, L., Silkina, A., Lupatsch, I., Grünewald, C. F., & Lovitt, R. (2018). Using microalgae in the circular economy to valorise anaerobic digestate: challenges and opportunities. *Bioresource Technology*, *267*, 732–742.

Stoffella J, P., & Kahn, B. A. (2001). *Compost utilization in horticultural cropping systems*. CRC Press, 1st edition.

Styles, D., Adams, P., Thelin, G., Vaneeckhaute, C., Chadwick, D., & Withers, P. J. A. (2018). Life cycle assessment of biofertilizer production and use compared with conventional liquid digestate management. *Environmental Science & Technology*, *52*(13), 7468–7476.

Taelman, S. E., de Meester, S., van Dijk, W., da Silva, V., & Dewulf, J. (2015). Environmental sustainability analysis of a protein-rich livestock feed ingredient in The Netherlands: Microalgae production versus soybean import. *Resources, Conservation and Recycling*, *101*, 61–72. https://doi.org/10.1016/J.RESCONREC.2015.05.013

Taha, R., Al-Rawas, A., Al-Jabri, K., Al-Harthy, A., Hassan, H., & Al-Oraimi, S. (2004). An overview of waste materials recycling in the Sultanate of Oman. *Resources, Conservation and Recycling*, *41*(4), 293–306.

Tampio, E., Marttinen, S., & Rintala, J. (2016). Liquid fertilizer products from anaerobic digestion of food waste: mass, nutrient and energy balance of four digestate liquid treatment systems. *Journal of Cleaner Production*, *125*, 22–32. https://doi.org/10.1016/J.JCLEPRO.2016.03.127

Tan, S. T., Ho, W. S., Hashim, H., Lee, C. T., Taib, M. R., & Ho, C. S. (2015). Energy, economic and environmental (3E) analysis of waste-to-energy (WTE) strategies for municipal solid waste (MSW) management in Malaysia. *Energy Conversion and Management*, *102*, 111–120. https://doi.org/10.1016/J.ENCONMAN.2015.02.010

Tao, W., Fattah, K. P., & Huchzermeier, M. P. (2016). Struvite recovery from anaerobically digested dairy manure: A review of application potential and hindrances. *Journal of Environmental Management*, *169*, 46–57. https://doi.org/10.1016/J.JENVMAN.2015.12.006

Tawfik, A., Eraky, M., Alhajeri, N. S., Ahmed, ·, Osman, I., David, ·, & Rooney, W. (2022). Cultivation of microalgae on liquid anaerobic digestate for depollution, biofuels and cosmetics: a review. *Environmental Chemistry Letters 2022*, *1*, 1–26. https://doi.org/10.1007/S10311-022-01481-2

Tchobanoglous, G., & Kreith, F. (2002). *Handbook of solid waste management*. McGraw-Hill Education.

Tchobanoglus, G., Burton, F., & Stensel, H. D. (2003). Wastewater engineering: Treatment and reuse. *American Water Works Association. Journal*, *95*(5), 201.

Teoh, S. K., & Li, L. Y. (2020). Feasibility of alternative sewage sludge treatment methods from a lifecycle assessment (LCA) perspective. *Journal of Cleaner Production*, *247*, 119495.

Theregowda, R. B., González-Mejía, A. M., Ma, X., & Garland, J. (2019). Nutrient Recovery from Municipal Wastewater for Sustainable Food Production Systems: An Alternative to Traditional Fertilizers. *Environmental Engineering Science*, *36*(7), 833–842. https://doi.org/10.1089/EES.2019.0053/ASSET/IMAGES/LARGE/FIGURE3.JPEG

Thompson, S., Sawyer, J., Bonam, R., & Valdivia, J. E. (2009). Building a better methane generation model: Validating models with methane recovery rates from 35 Canadian landfills. *Waste Management*, *29*(7), 2085–2091.

Tsapekos, P., Khoshnevisan, B., Alvarado-Morales, M., Zhu, X., Pan, J., Tian, H., & Angelidaki, I. (2021). Upcycling the anaerobic digestion streams in a bioeconomy approach: A review. *Renewable and Sustainable Energy Reviews*, *151*, 111635. https://doi.org/10.1016/J.RSER.2021.111635

Turovskiy, I. S., & Mathai, P. K. (2006). *Wastewater sludge processing*. John Wiley & Sons.

Udoeyo, F. F., & Hyee, A. (2002). Strengths of cement kiln dust concrete. *Journal of Materials in Civil Engineering*, *14*(6), 524–526.

US EPA. (n.d.). *Landfill Gas Emissions Model (LandGEM), Version 3.03*. https://www.epa.gov/catc/clean-air-technology-center-products#software

US EPA. (1999a). Biosolids generation, use, and disposal in the United States. *EPA530-R-99-009*.

US EPA. (1999b). *Characterization of Municipal Solid Waste in the United States: 1998 Update*.

US EPA. (2000). *Biosolids Technology Fact Sheet: Alkaline Stabilization of Biosolids*.

US EPA. (2002). *Biosolids Technology Fact Sheet: Use of Composting for Biosolids Management*. https://www.epa.gov/sites/production/files/2018-11/documents/use-composing-biosolids-management.pdf

US EPA. (2003). Environmental regulations and technology: Control of pathogens and vector attraction in sewage sludge. *United States Environment Protection Agency, Cincinnati, OH*.

US EPA. (2020). *Criteria for the Definition of Solid Waste and Solid and Hazardous Waste Exclusions*. https://www.epa.gov/hw/criteria-definition-solid-waste-and-solid-and-hazardous-waste-exclusions

USDOE. (2022). *EnergyPlus weather data*. U.S. Department of Energy's (DOE) Building Technologies Office (BTO), Managed by the National Renewable Energy Laboratory (NREL). https://energyplus.net/weather

Valderrama, C., Granados, R., & Cortina, J. L. (2013). Stabilisation of dewatered domestic sewage sludge by lime addition as raw material for the cement industry: understanding process and reactor performance. *Chemical Engineering Journal*, *232*, 458–467.

Valderrama, C., Granados, R., Cortina, J. L., Gasol, C. M., Guillem, M., & Josa, A. (2013). Comparative LCA of sewage sludge valorisation as both fuel and raw material substitute in clinker production. *Journal of Cleaner Production*, *51*, 205–213.

van Eekert, M. H. A., Weijma, J., Verdoes, N., de Buisonje, F. E., Reitsma, B. A. H., & van den Bulk, J. (2012). *Explorative research on innovative nitrogen recovery*. Stichting Toegepast Onderzoek Waterbeheer.

van Haaren, R., Themelis, N. J., & Barlaz, M. (2010). LCA comparison of windrow composting of yard wastes with use as alternative daily cover (ADC). *Waste Management*, *30*(12), 2649–2656. https://doi.org/10.1016/J.WASMAN.2010.06.007

van Velsen, A. F. M. (1979). Adaptation of methanogenic sludge to high ammonia-nitrogen concentrations. *Water Research*, *13*(10), 995–999. https://doi.org/10.1016/0043-1354(79)90194-5

Vaneeckhaute, C., Lebuf, V., Michels, E., Belia, E., Vanrolleghem, P. A., Tack, F. M. G., & Meers, E. (2017). Nutrient Recovery from Digestate: Systematic Technology Review and Product Classification. *Waste and Biomass Valorization*, *8*(1), 21–40. https://doi.org/10.1007/S12649-016-9642-X/TABLES/3

Vázquez-Rowe, I., Golkowska, K., Lebuf, V., Vaneeckhaute, C., Michels, E., Meers, E., Benetto, E., & Koster, D. (2015). Environmental assessment of digestate treatment technologies using LCA methodology. *Waste Management*, *43*, 442–459. https://doi.org/10.1016/J.WASMAN.2015.05.007

Vondra, M., Touš, M., & Teng, S. Y. (2019). Digestate evaporation treatment in biogas plants: A techno-economic assessment by Monte Carlo, neural networks and decision trees. *Journal of Cleaner Production*, *238*, 117870.

Wainaina, S., Awasthi, M. K., Sarsaiya, S., Chen, H., Singh, E., Kumar, A., Ravindran, B., Awasthi, S. K., Liu, T., & Duan, Y. (2020). Resource recovery and circular economy from organic solid waste using aerobic and anaerobic digestion technologies. *Bioresource Technology*, *301*, 122778.

Walker Industries. (2021). *N-Rich® Soil Amendment Safety Data Sheet*. https://www.walkerind.com/media/2020/01/N-Rich-SDS-2021.pdf

Walker, M., Iyer, K., Heaven, S., & Banks, C. J. (2011). Ammonia removal in anaerobic digestion by biogas stripping: An evaluation of process alternatives using a first order rate model based on experimental findings. *Chemical Engineering Journal*, *178*, 138–145. https://doi.org/10.1016/J.CEJ.2011.10.027

Wang, F., Fu, R., Lv, H., Zhu, G., Lu, B., Zhou, Z., Wu, X., & Chen, H. (2019). Phosphate Recovery from Swine Wastewater by a Struvite Precipitation Electrolyzer. *Scientific Reports 2019 9:1*, *9*(1), 1–10. https://doi.org/10.1038/s41598-019-45085-3

Wang, H., Brown, S. L., Magesan, G. N., Slade, A. H., Quintern, M., Clinton, P. W., & Payn, T. W. (2008). Technological options for the management of biosolids. *Environmental Science and Pollution Research*, *15*(4), 308–317. https://doi.org/10.1007/S11356-008-0012-5/FIGURES/1

Wang, L. K., Shammas, N. K., & Hung, Y.-T. (2007). *Biosolids treatment processes*. Humana Press Inc.

Wang, L. K., Shammas, N. K., & Hung, Y.-T. (2008). Biosolids Engineering and Management, Volume 7: Biosolids Engineering and Management. In *Biosolids Engineering and Management: Vol. Voulme 7*. Humana Press. https://doi.org/10.1007/978-1-59745-174-1

Wang, Q., Prasad, R., & Higgins, B. T. (2019). Aerobic bacterial pretreatment to overcome algal growth inhibition on high-strength anaerobic digestates. *Water Research*, *162*, 420–426.

Wang, Q., Wei, W., Liu, S., Yan, M., Song, K., Mai, J., Sun, J., Ni, B. J., & Gong, Y. (2018). Free Ammonia Pretreatment Improves Degradation of Secondary Sludge during Aerobic Digestion. *ACS Sustainable Chemistry and Engineering*, *6*(1), 1105–1111. https://doi.org/10.1021/ACSSUSCHEMENG.7B03403/ASSET/IMAGES/LARGE/SC-2017-03403T_0003.JPEG

Ward, A. J., Hobbs, P. J., Holliman, P. J., & Jones, D. L. (2008). Optimisation of the anaerobic digestion of agricultural resources. *Bioresource Technology*, *99*(17), 7928–7940. https://doi.org/10.1016/J.BIORTECH.2008.02.044

weatherspark. (2022). *Halifax Climate, Weather By Month, Average Temperature (Canada) - Weather Spark*. https://weatherspark.com/y/28434/Average-Weather-in-Halifax-Canada-Year-Round#Figures-Temperature

WEF. (2021). Energy in Water Resource Recovery Facilities - WEF Manual of Practice No. 32 (2nd Edition). In *Water Environment Federation (WEF)*. Water Environment Federation (WEF). https://app.knovel.com/hotlink/toc/id:kpU0UDEZC1/energy-in-water-resource/energy-in-water-resource

Weligama Thuppahige, R. T., & Babel, S. (2022). Environmental impact assessment of organic fraction of municipal solid waste treatment by anaerobic digestion in Sri Lanka. *Waste Management and Research*, *40*(2), 236–243. https://doi.org/10.1177/0734242X211013405

Wu, H., & Vaneeckhaute, C. (2022). Nutrient recovery from wastewater: A review on the integrated Physicochemical technologies of ammonia stripping, adsorption and struvite precipitation. *Chemical Engineering Journal*, *433*, 133664. https://doi.org/10.1016/J.CEJ.2021.133664

Wu, S., Ni, P., Li, J., Sun, H., Wang, Y., Luo, H., Dach, J., & Dong, R. (2016). Integrated approach to sustain biogas production in anaerobic digestion of chicken manure under recycled utilization of liquid digestate: Dynamics of ammonium accumulation and mitigation control. *Bioresource Technology*, *205*, 75–81. https://doi.org/10.1016/J.BIORTECH.2016.01.021

Xu, J., Zhao, Y., Zhao, G., & Zhang, H. (2015). Nutrient removal and biogas upgrading by integrating freshwater algae cultivation with piggery anaerobic digestate liquid treatment. *Applied Microbiology and Biotechnology*, *99*(15), 6493–6501.

Yen, H. W., & Brune, D. E. (2007). Anaerobic co-digestion of algal sludge and waste paper to produce methane. *Bioresource Technology*, *98*(1), 130–134. https://doi.org/10.1016/J.BIORTECH.2005.11.010

Yu, D., Kurola, J. M., Lähde, K., Kymäläinen, M., Sinkkonen, A., & Romantschuk, M. (2014). Biogas production and methanogenic archaeal community in mesophilic and thermophilic anaerobic co-digestion processes. *Journal of Environmental Management*, *143*, 54–60. https://doi.org/10.1016/J.JENVMAN.2014.04.025

Zahan, Z., Othman, M. Z., & Muster, T. H. (2018). Anaerobic digestion/co-digestion kinetic potentials of different agro-industrial wastes: A comparative batch study for C/N optimisation. *Waste Management*, *71*, 663–674. https://doi.org/10.1016/J.WASMAN.2017.08.014

Zamanzadeh, M., Hagen, L. H., Svensson, K., Linjordet, R., & Horn, S. J. (2016). Anaerobic digestion of food waste – Effect of recirculation and temperature on performance and microbiology. *Water Research*, *96*, 246–254. https://doi.org/10.1016/J.WATRES.2016.03.058

Zarebska, A., Romero Nieto, D., Christensen, K. V., Fjerbæk Søtoft, L., & Norddahl, B. (2015). Ammonium fertilizers production from manure: a critical review. *Critical Reviews in Environmental Science and Technology*, *45*(14), 1469–1521.

Zhang, Y., Banks, C. J., & Heaven, S. (2012). Co-digestion of source segregated domestic food waste to improve process stability. *Bioresource Technology*, *114*, 168–178. https://doi.org/10.1016/J.BIORTECH.2012.03.040

Zisopoulos, F. K., van der Goot, A. J., & Boom, R. M. (2018). Exergy destruction in ammonia scrubbers. *Resources, Conservation and Recycling*, *136*, 153–165. https://doi.org/10.1016/J.RESCONREC.2018.04.014

Appendix A: wastewater treatment inventory

Table E-1 Summary of WWT process inventory, adopted from ecoinvent V3 database. The functional unit for this process is treatment of 1 m^3 wastewater.

Flow	Amount	Unit
input		
aluminium sulfate, powder	0.003151	kg
ammonia, liquid	3.36E-05	kg
cement, unspecified	6.91E-04	kg
chemical, inorganic	3.07E-07	kg
chemical, organic	2.45E-07	kg
chromium oxide, flakes	1.96E-08	kg
electricity, high voltage	8.69E-03	kWh
electricity, low voltage	0.20571	kWh
heat, district or industrial, natural gas	0.00295	MJ
heat, district or industrial, other than natural gas	0.05072	MJ
hydrochloric acid, without water, in 30% solution state	1.84E-07	kg
iron (III) chloride, without water, in 40% solution state	0.015944	kg
iron sulfate	0.01166	kg
liquid manure spreading, by vacuum tanker	6.04E-04	m3
municipal waste incineration facility	1.51E-11	Item(s)
process-specific burdens, municipal waste incineration	0.060476	kg
process-specific burdens, residual material landfill	0.001727	kg
process-specific burdens, slag landfill	0.011013	kg
quicklime, milled, packed	5.85E-07	kg
residual material landfill	3.60E-12	Item(s)
sewer grid, 1E9l/year, 30 km	2.82E-07	km
slag landfill	1.96E-11	Item(s)
sodium hydroxide, without water, in 50% solution state	1.64E-04	kg
titanium dioxide	9.62E-07	kg
waste cement, hydrated	-0.00173	kg
waste graphical paper	-0.0155	kg
waste plastic, mixture	-1.55E-02	kg
wastewater treatment facility, capacity 1E9l/year	2.66E-08	Item(s)
Output		
AluminiumElementary flows/Emission to air/high population density	6.58E-07	kg
AluminiumElementary flows/Emission to soil/agricultural	0.001052	kg
AluminiumElementary flows/Emission to water/ground water, long-term	3.12E-04	kg
AluminiumElementary flows/Emission to water/surface water	6.23E-05	kg

AmmoniaElementary flows/Emission to air/high population density	5.94E-04	kg
Ammonium, ionElementary flows/Emission to water/surface water	0.011027	kg
ArsenicElementary flows/Emission to air/high population density	2.53E-10	kg
ArsenicElementary flows/Emission to soil/agricultural	1.39E-07	kg
Arsenic, ionElementary flows/Emission to water/ground water, long-term	3.05E-08	kg
Arsenic, ionElementary flows/Emission to water/surface water	7.31E-07	kg
BOD5, Biological Oxygen DemandElementary flows/Emission to water/ground water, long-term	3.99E-05	kg
BOD5, Biological Oxygen DemandElementary flows/Emission to water/surface water	0.009823	kg
CadmiumElementary flows/Emission to air/high population density	2.24E-12	kg
CadmiumElementary flows/Emission to soil/agricultural	9.87E-08	kg
Cadmium, ionElementary flows/Emission to water/ground water, long-term	3.97E-10	kg
Cadmium, ionElementary flows/Emission to water/surface water	1.42E-07	kg
CalciumElementary flows/Emission to air/high population density	2.38E-06	kg
CalciumElementary flows/Emission to soil/agricultural	0.003561	kg
Calcium, ionElementary flows/Emission to water/ground water, long-term	0.001241	kg
Calcium, ionElementary flows/Emission to water/surface water	0.045852	kg
CarbonElementary flows/Emission to soil/agricultural	0.012346	kg
Carbon dioxide, non-fossilElementary flows/Emission to air/high population density	0.17202	kg
Carbon monoxide, non-fossilElementary flows/Emission to air/high population density	1.61E-04	kg
ChlorideElementary flows/Emission to water/surface water	0.040484	kg
ChromiumElementary flows/Emission to air/high population density	1.27E-13	kg
ChromiumElementary flows/Emission to soil/agricultural	4.30E-06	kg
Chromium VIElementary flows/Emission to water/ground water, long-term	1.82E-07	kg
Chromium VIElementary flows/Emission to water/surface water	6.26E-06	kg
Chromium, ionElementary flows/Emission to water/surface water	5.49E-09	kg
CobaltElementary flows/Emission to air/high population density	7.24E-15	kg
CobaltElementary flows/Emission to soil/agricultural	5.69E-07	kg
CobaltElementary flows/Emission to water/ground water, long-term	2.00E-07	kg
CobaltElementary flows/Emission to water/surface water	8.21E-07	kg
COD, Chemical Oxygen DemandElementary flows/Emission to water/ground water, long-term	1.22E-04	kg
COD, Chemical Oxygen DemandElementary flows/Emission to water/surface water	0.030152	kg
CopperElementary flows/Emission to air/high population density	5.85E-11	kg
CopperElementary flows/Emission to soil/agricultural	1.98E-05	kg
Copper, ionElementary flows/Emission to water/ground water, long-term	6.41E-06	kg
Copper, ionElementary flows/Emission to water/surface water	9.71E-06	kg
CyanideElementary flows/Emission to air/high population density	6.00E-07	kg

Dinitrogen monoxideElementary flows/Emission to air/high population density	1.15E-04	kg
DOC, Dissolved Organic CarbonElementary flows/Emission to water/ground water, long-term	4.83E-05	kg
DOC, Dissolved Organic CarbonElementary flows/Emission to water/surface water	0.007538	kg
FluorideElementary flows/Emission to water/surface water	3.28E-05	kg
IronElementary flows/Emission to air/high population density	1.27E-07	kg
IronElementary flows/Emission to soil/agricultural	0.009475	kg
Iron, ionElementary flows/Emission to water/ground water, long-term	0.001778	kg
Iron, ionElementary flows/Emission to water/surface water	0.003601	kg
LeadElementary flows/Emission to air/high population density	8.18E-11	kg
LeadElementary flows/Emission to soil/agricultural	5.49E-06	kg
LeadElementary flows/Emission to water/ground water, long-term	1.57E-07	kg
LeadElementary flows/Emission to water/surface water	9.49E-07	kg
MagnesiumElementary flows/Emission to air/high population density	2.21E-07	kg
MagnesiumElementary flows/Emission to soil/agricultural	4.00E-04	kg
MagnesiumElementary flows/Emission to water/ground water, long-term	1.48E-04	kg
MagnesiumElementary flows/Emission to water/surface water	0.005148	kg
ManganeseElementary flows/Emission to air/high population density	4.07E-14	kg
ManganeseElementary flows/Emission to soil/agricultural	1.86E-05	kg
ManganeseElementary flows/Emission to water/ground water, long-term	6.43E-06	kg
ManganeseElementary flows/Emission to water/surface water	2.69E-05	kg
MercuryElementary flows/Emission to air/high population density	3.35E-13	kg
MercuryElementary flows/Emission to soil/agricultural	9.87E-08	kg
MercuryElementary flows/Emission to water/ground water, long-term	2.06E-09	kg
MercuryElementary flows/Emission to water/surface water	6.23E-08	kg
Methane, non-fossilElementary flows/Emission to air/high population density	5.02E-04	kg
MolybdenumElementary flows/Emission to air/high population density	2.70E-10	kg
MolybdenumElementary flows/Emission to soil/agricultural	3.37E-07	kg
MolybdenumElementary flows/Emission to water/ground water, long-term	1.12E-07	kg
MolybdenumElementary flows/Emission to water/surface water	5.09E-07	kg
NickelElementary flows/Emission to air/high population density	3.20E-14	kg
NickelElementary flows/Emission to soil/agricultural	1.85E-06	kg
Nickel, ionElementary flows/Emission to water/ground water, long-term	6.94E-07	kg
Nickel, ionElementary flows/Emission to water/surface water	4.00E-06	kg
NitrateElementary flows/Emission to water/ground water, long-term	2.39E-05	kg
NitrateElementary flows/Emission to water/surface water	0.048295	kg
NitriteElementary flows/Emission to water/surface water	6.44E-04	kg
NitrogenElementary flows/Emission to water/surface water	4.90E-04	kg

Nitrogen oxidesElementary flows/Emission to air/high population density	6.76E-04	kg
NMVOC, non-methane volatile organic compounds, unspecified originElementary flows/Emission to air/high population density	2.28E-06	kg
PhosphateElementary flows/Emission to water/ground water	2.71E-05	kg
PhosphateElementary flows/Emission to water/ground water, long-term	7.27E-05	kg
PhosphateElementary flows/Emission to water/surface water	0.002748	kg
PhosphorusElementary flows/Emission to air/high population density	6.20E-07	kg
Potassium, ionElementary flows/Emission to water/surface water	3.99E-04	kg
SiliconElementary flows/Emission to air/high population density	1.96E-06	kg
SiliconElementary flows/Emission to soil/agricultural	0.002098	kg
SiliconElementary flows/Emission to water/ground water, long-term	7.30E-05	kg
SiliconElementary flows/Emission to water/surface water	1.88E-04	kg
Sodium, ionElementary flows/Emission to water/surface water	0.002186	kg
SulfateElementary flows/Emission to water/ground water, long-term	0.001104	kg
SulfateElementary flows/Emission to water/surface water	0.14465	kg
SulfurElementary flows/Emission to soil/agricultural	0.001099	kg
Sulfur dioxideElementary flows/Emission to air/high population density	8.84E-04	kg
TinElementary flows/Emission to air/high population density	7.51E-10	kg
TinElementary flows/Emission to soil/agricultural	1.41E-06	kg
Tin, ionElementary flows/Emission to water/ground water, long-term	2.84E-07	kg
Tin, ionElementary flows/Emission to water/surface water	1.42E-06	kg
TOC, Total Organic CarbonElementary flows/Emission to water/ground water, long-term	4.83E-05	kg
TOC, Total Organic CarbonElementary flows/Emission to water/surface water	0.007299	kg
wastewater, average	-1	m3
WaterElementary flows/Emission to air/unspecified	0.1	m3
WaterElementary flows/Emission to water/unspecified	0.9	m3
ZincElementary flows/Emission to air/high population density	3.53E-10	kg
ZincElementary flows/Emission to soil/agricultural	5.39E-05	kg
Zinc, ionElementary flows/Emission to water/ground water, long-term	3.35E-07	kg
Zinc, ionElementary flows/Emission to water/surface water	3.38E-05	kg

Appendix B: LandGem Results

Table B-1 Landgem results for otter lake MSOF stream

Year	Residual LFG							
	Total landfill gas		Methane		Carbon dioxide		NMOC	
	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)
2019	0	0	0	0	0	0	0	0
2020	28.72251	22999.65	7.672091	11499.83	21.05042	11499.83	0.049465	13.79979
2021	27.15826	21747.07	7.254263	10873.54	19.904	10873.54	0.046771	13.04824
2022	25.6792	20562.71	6.859189	10281.35	18.82001	10281.35	0.044224	12.33763
2023	24.28069	19442.85	6.485632	9721.423	17.79505	9721.423	0.041815	11.66571
2024	22.95834	18383.97	6.132419	9191.986	16.82592	9191.986	0.039538	11.03038
2025	21.70801	17382.77	5.798442	8691.383	15.90957	8691.383	0.037385	10.42966
2026	20.52577	16436.08	5.482654	8218.042	15.04312	8218.042	0.035349	9.861651
2027	19.40792	15540.96	5.184064	7770.481	14.22386	7770.481	0.033424	9.324577
2028	18.35095	14694.59	4.901735	7347.294	13.44921	7347.294	0.031603	8.816752
2029	17.35154	13894.31	4.634782	6947.154	12.71676	6947.154	0.029882	8.336584
2030	16.40656	13137.61	4.382368	6568.806	12.02419	6568.806	0.028255	7.882567
2031	15.51304	12422.13	4.143701	6211.063	11.36934	6211.063	0.026716	7.453275
2032	14.66819	11745.61	3.918031	5872.803	10.75016	5872.803	0.025261	7.047364
2033	13.86935	11105.93	3.704652	5552.965	10.1647	5552.965	0.023885	6.663558
2034	13.11401	10501.09	3.502893	5250.546	9.611118	5250.546	0.022584	6.300655
2035	12.39981	9929.193	3.312123	4964.597	9.087688	4964.597	0.021355	5.957516
2036	11.72451	9388.441	3.131742	4694.22	8.592765	4694.22	0.020192	5.633064
2037	11.08598	8877.138	2.961184	4438.569	8.124795	4438.569	0.019092	5.326283
2038	10.48223	8393.681	2.799916	4196.841	7.682312	4196.841	0.018052	5.036209
2039	9.911356	7936.554	2.64743	3968.277	7.263927	3968.277	0.017069	4.761933
2040	9.371575	7504.323	2.503248	3752.161	6.868327	3752.161	0.016139	4.502594
2041	8.861191	7095.631	2.366919	3547.815	6.494272	3547.815	0.01526	4.257378
2042	8.378603	6709.197	2.238015	3354.598	6.140588	3354.598	0.014429	4.025518
2043	7.922297	6343.808	2.116131	3171.904	5.806166	3171.904	0.013644	3.806285
2044	7.490842	5998.319	2.000884	2999.159	5.489958	2999.159	0.0129	3.598991
2045	7.082884	5671.645	1.891914	2835.823	5.19097	2835.823	0.012198	3.402987
2046	6.697144	5362.762	1.788879	2681.381	4.908265	2681.381	0.011534	3.217657
2047	6.332412	5070.702	1.691455	2535.351	4.640957	2535.351	0.010905	3.042421
2048	5.987543	4794.547	1.599337	2397.273	4.388206	2397.273	0.010312	2.876728
2049	5.661457	4533.432	1.512236	2266.716	4.149221	2266.716	0.00975	2.720059
2050	5.353129	4286.537	1.429878	2143.269	3.923251	2143.269	0.009219	2.571922
2051	5.061593	4053.089	1.352006	2026.544	3.709587	2026.544	0.008717	2.431853
2052	4.785934	3832.354	1.278374	1916.177	3.50756	1916.177	0.008242	2.299412

2053	4.525288	3623.641	1.208753	1811.82	3.316535	1811.82	0.007793	2.174184
2054	4.278837	3426.294	1.142923	1713.147	3.135914	1713.147	0.007369	2.055776
2055	4.045808	3239.695	1.080679	1619.848	2.965129	1619.848	0.006968	1.943817
2056	3.825469	3063.258	1.021824	1531.629	2.803645	1531.629	0.006588	1.837955
2057	3.617131	2896.431	0.966175	1448.215	2.650957	1448.215	0.006229	1.737858
2058	3.420139	2738.689	0.913556	1369.344	2.506583	1369.344	0.00589	1.643213
2059	3.233875	2589.537	0.863803	1294.769	2.370072	1294.769	0.005569	1.553722
2060	3.057756	2448.509	0.816759	1224.254	2.240996	1224.254	0.005266	1.469105
2061	2.891228	2315.161	0.772278	1157.58	2.11895	1157.58	0.004979	1.389097
2062	2.733769	2189.075	0.730219	1094.538	2.00355	1094.538	0.004708	1.313445
2063	2.584885	2069.856	0.690451	1034.928	1.894435	1034.928	0.004452	1.241914
2064	2.44411	1957.13	0.652848	978.5651	1.791262	978.5651	0.004209	1.174278
2065	2.311002	1850.543	0.617293	925.2716	1.693709	925.2716	0.00398	1.110326
2066	2.185143	1749.761	0.583675	874.8805	1.601468	874.8805	0.003763	1.049857
2067	2.066138	1654.468	0.551888	827.2338	1.51425	827.2338	0.003558	0.992681
2068	1.953614	1564.364	0.521831	782.1819	1.431783	782.1819	0.003364	0.938618
2069	1.847219	1479.167	0.493412	739.5836	1.353807	739.5836	0.003181	0.8875
2070	1.746618	1398.61	0.46654	699.3052	1.280077	699.3052	0.003008	0.839166
2071	1.651495	1322.441	0.441132	661.2205	1.210363	661.2205	0.002844	0.793465
2072	1.561554	1250.42	0.417108	625.2098	1.144446	625.2098	0.002689	0.750252
2073	1.47651	1182.321	0.394392	591.1604	1.082118	591.1604	0.002543	0.709392
2074	1.396098	1117.931	0.372913	558.9653	1.023185	558.9653	0.002404	0.670758
2075	1.320065	1057.047	0.352604	528.5235	0.967462	528.5235	0.002273	0.634228
2076	1.248173	999.4794	0.333401	499.7397	0.914773	499.7397	0.00215	0.599688
2077	1.180197	945.0468	0.315243	472.5234	0.864954	472.5234	0.002032	0.567028
2078	1.115922	893.5788	0.298075	446.7894	0.817847	446.7894	0.001922	0.536147
2079	1.055148	844.9137	0.281841	422.4569	0.773307	422.4569	0.001817	0.506948
2080	0.997684	798.899	0.266492	399.4495	0.731192	399.4495	0.001718	0.479339
2081	0.943349	755.3902	0.251979	377.6951	0.69137	377.6951	0.001625	0.453234
2082	0.891974	714.251	0.238256	357.1255	0.653718	357.1255	0.001536	0.428551
2083	0.843396	675.3523	0.22528	337.6762	0.618116	337.6762	0.001452	0.405211
2084	0.797464	638.572	0.213011	319.286	0.584453	319.286	0.001373	0.383143
2085	0.754033	603.7949	0.20141	301.8974	0.552623	301.8974	0.001299	0.362277
2086	0.712968	570.9117	0.190441	285.4558	0.522527	285.4558	0.001228	0.342547
2087	0.674139	539.8193	0.18007	269.9097	0.494069	269.9097	0.001161	0.323892
2088	0.637425	510.4203	0.170263	255.2101	0.467162	255.2101	0.001098	0.306252
2089	0.60271	482.6224	0.16099	241.3112	0.44172	241.3112	0.001038	0.289573
2090	0.569886	456.3383	0.152223	228.1692	0.417663	228.1692	0.000981	0.273803
2091	0.53885	431.4858	0.143933	215.7429	0.394917	215.7429	0.000928	0.258891
2092	0.509503	407.9867	0.136094	203.9933	0.37341	203.9933	0.000877	0.244792

2093	0.481755	385.7674	0.128682	192.8837	0.353073	192.8837	0.00083	0.23146
2094	0.455519	364.7581	0.121674	182.3791	0.333845	182.3791	0.000784	0.218855
2095	0.430711	344.8931	0.115047	172.4465	0.315663	172.4465	0.000742	0.206936
2096	0.407254	326.1099	0.108782	163.055	0.298472	163.055	0.000701	0.195666
2097	0.385074	308.3497	0.102857	154.1748	0.282217	154.1748	0.000663	0.18501
2098	0.364103	291.5567	0.097256	145.7783	0.266847	145.7783	0.000627	0.174934
2099	0.344274	275.6783	0.091959	137.8391	0.252314	137.8391	0.000593	0.165407
2100	0.325524	260.6646	0.086951	130.3323	0.238573	130.3323	0.000561	0.156399
2101	0.307796	246.4686	0.082216	123.2343	0.22558	123.2343	0.00053	0.147881
2102	0.291033	233.0457	0.077738	116.5228	0.213295	116.5228	0.000501	0.139827
2103	0.275183	220.3538	0.073504	110.1769	0.201679	110.1769	0.000474	0.132212
2104	0.260196	208.3532	0.069501	104.1766	0.190695	104.1766	0.000448	0.125012
2105	0.246026	197.0061	0.065716	98.50303	0.18031	98.50303	0.000424	0.118204
2106	0.232627	186.2769	0.062137	93.13847	0.17049	93.13847	0.000401	0.111766
2107	0.219958	176.1321	0.058753	88.06607	0.161205	88.06607	0.000379	0.105679
2108	0.207979	166.5398	0.055553	83.26991	0.152425	83.26991	0.000358	0.099924
2109	0.196652	157.4699	0.052528	78.73496	0.144124	78.73496	0.000339	0.094482
2110	0.185942	148.894	0.049667	74.44699	0.136275	74.44699	0.00032	0.089336
2111	0.175816	140.7851	0.046962	70.39254	0.128853	70.39254	0.000303	0.084471
2112	0.166241	133.1178	0.044405	66.5589	0.121836	66.5589	0.000286	0.079871
2113	0.157187	125.8681	0.041986	62.93405	0.115201	62.93405	0.000271	0.075521
2114	0.148627	119.0132	0.0397	59.5066	0.108927	59.5066	0.000256	0.071408
2115	0.140532	112.5316	0.037538	56.26582	0.102995	56.26582	0.000242	0.067519
2116	0.132879	106.4031	0.035493	53.20154	0.097385	53.20154	0.000229	0.063842
2117	0.125642	100.6083	0.03356	50.30414	0.092082	50.30414	0.000216	0.060365
2118	0.118799	95.12906	0.031733	47.56453	0.087067	47.56453	0.000205	0.057077

Table B-2 Summary of landGem results for residual stream in otter lake

Year	Stabilized LFG							
	Total landfill gas		Methane		Carbon dioxide		NMOC	
	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)	(Mg/year)	(m3/year)
2019	0	0	0	0	0	0	0	0
2020	22.54779	18055.22	6.022756	9027.611	16.52503	9027.611	0.038831	10.83313
2021	21.31982	17071.92	5.694752	8535.96	15.62506	8535.96	0.036716	10.24315
2022	20.15872	16142.17	5.384611	8071.084	14.77411	8071.084	0.034717	9.685301
2023	19.06086	15263.05	5.09136	7631.526	13.9695	7631.526	0.032826	9.157831
2024	18.02279	14431.81	4.81408	7215.906	13.20871	7215.906	0.031038	8.659087
2025	17.04125	13645.84	4.551901	6822.922	12.48935	6822.922	0.029348	8.187506
2026	16.11317	12902.68	4.304001	6451.339	11.80917	6451.339	0.02775	7.741607

2027	15.23563	12199.99	4.069601	6099.994	11.16603	6099.994	0.026238	7.319993
2028	14.40589	11535.57	3.847967	5767.783	10.55792	5767.783	0.024809	6.92134
2029	13.62133	10907.33	3.638404	5453.665	9.982926	5453.665	0.023458	6.544397
2030	12.8795	10313.31	3.440253	5156.653	9.439248	5156.653	0.022181	6.187984
2031	12.17807	9751.635	3.252894	4875.817	8.925178	4875.817	0.020973	5.850981
2032	11.51484	9220.552	3.075738	4610.276	8.439105	4610.276	0.01983	5.532331
2033	10.88774	8718.393	2.908231	4359.197	7.979504	4359.197	0.01875	5.231036
2034	10.29478	8243.582	2.749846	4121.791	7.544933	4121.791	0.017729	4.946149
2035	9.734117	7794.629	2.600087	3897.315	7.13403	3897.315	0.016764	4.676778
2036	9.203989	7370.127	2.458484	3685.064	6.745504	3685.064	0.015851	4.422076
2037	8.702731	6968.744	2.324593	3484.372	6.378138	3484.372	0.014988	4.181246
2038	8.228773	6589.22	2.197994	3294.61	6.030779	3294.61	0.014171	3.953532
2039	7.780627	6230.365	2.078289	3115.183	5.702338	3115.183	0.0134	3.738219
2040	7.356887	5891.054	1.965104	2945.527	5.391784	2945.527	0.01267	3.534632
2041	6.956225	5570.222	1.858082	2785.111	5.098143	2785.111	0.01198	3.342133
2042	6.577383	5266.863	1.75689	2633.432	4.820493	2633.432	0.011327	3.160118
2043	6.219173	4980.025	1.661208	2490.013	4.557965	2490.013	0.01071	2.988015
2044	5.880471	4708.809	1.570737	2354.404	4.309734	2354.404	0.010127	2.825285
2045	5.560216	4452.363	1.485193	2226.181	4.075022	2226.181	0.009576	2.671418
2046	5.257402	4209.883	1.404308	2104.942	3.853093	2104.942	0.009054	2.52593
2047	4.971079	3980.61	1.327829	1990.305	3.64325	1990.305	0.008561	2.388366
2048	4.70035	3763.822	1.255514	1881.911	3.444836	1881.911	0.008095	2.258293
2049	4.444365	3558.841	1.187138	1779.421	3.257227	1779.421	0.007654	2.135305
2050	4.202321	3365.024	1.122485	1682.512	3.079836	1682.512	0.007237	2.019014
2051	3.973459	3181.761	1.061354	1590.881	2.912105	1590.881	0.006843	1.909057
2052	3.757061	3008.48	1.003551	1504.24	2.753509	1504.24	0.00647	1.805088
2053	3.552448	2844.636	0.948897	1422.318	2.603551	1422.318	0.006118	1.706781
2054	3.358979	2689.714	0.897219	1344.857	2.461759	1344.857	0.005785	1.613829
2055	3.176046	2543.23	0.848356	1271.615	2.32769	1271.615	0.00547	1.525938
2056	3.003076	2404.724	0.802154	1202.362	2.200922	1202.362	0.005172	1.442834
2057	2.839525	2273.76	0.758468	1136.88	2.081058	1136.88	0.00489	1.364256
2058	2.684882	2149.929	0.717161	1074.965	1.967721	1074.965	0.004624	1.289958
2059	2.538661	2032.842	0.678104	1016.421	1.860558	1016.421	0.004372	1.219705
2060	2.400404	1922.132	0.641174	961.066	1.75923	961.066	0.004134	1.153279
2061	2.269676	1817.451	0.606255	908.7255	1.663421	908.7255	0.003909	1.090471
2062	2.146067	1718.471	0.573238	859.2355	1.57283	859.2355	0.003696	1.031083
2063	2.02919	1624.882	0.542019	812.4408	1.487172	812.4408	0.003495	0.974929
2064	1.918679	1536.389	0.5125	768.1946	1.406179	768.1946	0.003304	0.921833
2065	1.814186	1452.716	0.484589	726.358	1.329598	726.358	0.003124	0.87163
2066	1.715384	1373.6	0.458197	686.7999	1.257186	686.7999	0.002954	0.82416

2067	1.621963	1298.792	0.433244	649.3962	1.188719	649.3962	0.002793	0.779275
2068	1.533629	1228.059	0.409649	614.0295	1.12398	614.0295	0.002641	0.736835
2069	1.450106	1161.178	0.387339	580.589	1.062767	580.589	0.002497	0.696707
2070	1.371132	1097.939	0.366244	548.9696	1.004888	548.9696	0.002361	0.658764
2071	1.296459	1038.144	0.346298	519.0722	0.950161	519.0722	0.002233	0.622887
2072	1.225853	981.6062	0.327439	490.8031	0.898415	490.8031	0.002111	0.588964
2073	1.159092	928.1471	0.309606	464.0735	0.849486	464.0735	0.001996	0.556888
2074	1.095967	877.5994	0.292745	438.7997	0.803222	438.7997	0.001887	0.52656
2075	1.03628	829.8046	0.276801	414.9023	0.759478	414.9023	0.001785	0.497883
2076	0.979843	784.6127	0.261727	392.3064	0.718116	392.3064	0.001687	0.470768
2077	0.92648	741.882	0.247473	370.941	0.679007	370.941	0.001596	0.445129
2078	0.876023	701.4785	0.233995	350.7392	0.642028	350.7392	0.001509	0.420887
2079	0.828314	663.2754	0.221252	331.6377	0.607062	331.6377	0.001426	0.397965
2080	0.783203	627.1528	0.209202	313.5764	0.574001	313.5764	0.001349	0.376292
2081	0.740549	592.9975	0.197809	296.4988	0.542741	296.4988	0.001275	0.355799
2082	0.700218	560.7024	0.187036	280.3512	0.513183	280.3512	0.001206	0.336421
2083	0.662084	530.166	0.17685	265.083	0.485234	265.083	0.00114	0.3181
2084	0.626026	501.2927	0.167218	250.6464	0.458808	250.6464	0.001078	0.300776
2085	0.591932	473.9919	0.158111	236.9959	0.433821	236.9959	0.001019	0.284395
2086	0.559695	448.1779	0.149501	224.0889	0.410195	224.0889	0.000964	0.268907
2087	0.529214	423.7697	0.141359	211.8849	0.387855	211.8849	0.000911	0.254262
2088	0.500392	400.6909	0.13366	200.3454	0.366732	200.3454	0.000862	0.240415
2089	0.47314	378.8689	0.126381	189.4344	0.34676	189.4344	0.000815	0.227321
2090	0.447373	358.2354	0.119498	179.1177	0.327875	179.1177	0.00077	0.214941
2091	0.423008	338.7256	0.11299	169.3628	0.310018	169.3628	0.000728	0.203235
2092	0.399971	320.2783	0.106837	160.1391	0.293134	160.1391	0.000689	0.192167
2093	0.378188	302.8356	0.101018	151.4178	0.27717	151.4178	0.000651	0.181701
2094	0.357592	286.343	0.095517	143.1715	0.262075	143.1715	0.000616	0.171806
2095	0.338117	270.7485	0.090315	135.3742	0.247802	135.3742	0.000582	0.162449
2096	0.319703	256.0033	0.085396	128.0016	0.234307	128.0016	0.000551	0.153602
2097	0.302292	242.0611	0.080745	121.0306	0.221546	121.0306	0.000521	0.145237
2098	0.285829	228.8783	0.076348	114.4391	0.209481	114.4391	0.000492	0.137327
2099	0.270262	216.4133	0.07219	108.2067	0.198072	108.2067	0.000465	0.129848
2100	0.255543	204.6273	0.068258	102.3136	0.187285	102.3136	0.00044	0.122776
2101	0.241626	193.4831	0.064541	96.74156	0.177085	96.74156	0.000416	0.11609
2102	0.228467	182.9459	0.061026	91.47293	0.167441	91.47293	0.000393	0.109768
2103	0.216025	172.9825	0.057702	86.49123	0.158322	86.49123	0.000372	0.103789
2104	0.20426	163.5617	0.05456	81.78085	0.1497	81.78085	0.000352	0.098137
2105	0.193136	154.654	0.051589	77.32699	0.141547	77.32699	0.000333	0.092792
2106	0.182617	146.2314	0.048779	73.1157	0.133838	73.1157	0.000314	0.087739

2107	0.172672	138.2675	0.046122	69.13375	0.126549	69.13375	0.000297	0.082961
2108	0.163268	130.7373	0.043611	65.36867	0.119657	65.36867	0.000281	0.078442
2109	0.154376	123.6173	0.041236	61.80863	0.113141	61.80863	0.000266	0.07417
2110	0.145969	116.885	0.03899	58.44248	0.106979	58.44248	0.000251	0.070131
2111	0.138019	110.5193	0.036866	55.25965	0.101153	55.25965	0.000238	0.066312
2112	0.130502	104.5003	0.034859	52.25017	0.095644	52.25017	0.000225	0.0627
2113	0.123395	98.80915	0.03296	49.40458	0.090435	49.40458	0.000213	0.059285
2114	0.116675	93.42792	0.031165	46.71396	0.08551	46.71396	0.000201	0.056057
2115	0.110321	88.33975	0.029468	44.16988	0.080853	44.16988	0.00019	0.053004
2116	0.104313	83.5287	0.027863	41.76435	0.07645	41.76435	0.00018	0.050117
2117	0.098632	78.97965	0.026346	39.48983	0.072286	39.48983	0.00017	0.047388
2118	0.09326	74.67835	0.024911	37.33918	0.068349	37.33918	0.000161	0.044807

Appendix C: Waste Collection Regions

Table C-1 Summary of transportation inventory for HRM solid waste collection. Since no information is available on the destination of the source separated organcis (SSO), it is assumed that the collected stream is sent to the closet composting facility. The amount of collected waste in each district is estimated based on normalizing the total generated waste based on that specific district population.

District	Lat	Long	To Otter Lake, km	To Ragge Lake, km	To Miller, km	closest compost facility, km	Population	normalized garbage,t	normalized SSO, t	garbage transport, t*km	SSO transport, t*km
1	44.63139301	-63.42938393	36	31	22	22	1890	211	239	7606	5256
2	44.70794572	-63.53068579	26	20	7	7	5106	570	646	14567	4669
3	44.84232318	-62.97334282	113	113	98	98	5945	664	752	75127	73375
4	44.66666034	-63.59988628	16	12	11	11	3314	370	419	6022	4687
5	44.71248781	-64.02810485	40	44	54	44	5442	607	688	24236	30357
6	44.67205428	-63.58537603	19	14	8	8	3132	350	396	6738	3023
7	44.66755346	-63.56148011	20	15	9	9	4623	516	585	10314	5090
8	44.6058814	-63.5048119	30	24	18	18	4583	512	580	15138	10706
9	44.66913998	-63.66918173	12	9	18	9	4199	469	531	5836	4989
10	44.78575043	-63.68047948	29	28	18	18	4945	552	626	15898	11297
11	44.67940731	-63.64952974	15	11	16	11	3409	381	431	5532	4950
12	44.67630242	-63.57215871	19	14	8	8	3200	357	405	6920	3085
13	44.62891977	-63.6383845	13	7	16	7	3766	420	476	5429	3158
14	44.72893469	-63.63921592	22	20	15	15	3970	443	502	9843	7664
15	44.64005631	-63.56524971	19	14	12	12	3466	387	438	7238	5311
16	44.64467433	-63.58395967	17	12	12	12	1859	208	235	3485	2743
17	44.68345111	-63.50630774	26	21	10	10	4613	515	584	13240	6088
18	44.78923829	-63.70712717	26	28	21	21	6146	686	778	17865	15945
19	44.68574189	-63.56231047	21	16	8	8	4769	532	603	11433	4537
20	44.66764142	-63.48445785	27	22	13	13	3131	350	396	9534	5207
21	44.64125773	-63.64182661	12	7	16	7	2914	325	369	4060	2554
22	44.64187655	-63.61077661	14	9	13	9	2561	286	324	4077	2958
23	44.68843405	-63.54087518	23	18	7	7	3000	335	380	7659	2833
24	44.64543265	-63.47838047	29	24	15	15	5174	578	655	16997	9925
25	44.64462572	-63.56995338	18	13	11	11	2778	310	351	5664	4004
26	44.66736549	-63.57565865	19	14	8	8	1481	165	187	3122	1586
27	44.77791936	-63.79072241	28	30	31	30	5114	571	647	16009	19572
28	44.67414752	-63.54508438	22	17	9	9	4228	472	535	10331	4770
29	44.5461685	-63.82797355	23	22	42	22	6769	756	856	17534	19089
30	44.66343229	-63.66945661	12	9	18	9	3967	443	502	5214	4374

31	44.65137249	-63.57406546	18	13	11	11	2357	263	298	4780	3315
32	44.67974543	-63.49662077	27	22	11	11	3191	356	404	9518	4632
33	44.65627536	-63.66119497	11	8	17	8	8232	919	1041	10511	8727
34	44.65244897	-63.60514717	15	10	11	10	5062	565	640	8659	6526
35	44.59439309	-63.44042633	33	28	22	22	5624	628	711	21023	15911
36	44.65638511	-63.58479	17	12	10	10	5036	562	637	9535	6499
37	44.62576297	-63.56513802	19	14	15	14	2955	330	374	6329	5254
38	44.6448267	-63.53986177	23	18	12	12	3855	430	488	10067	5983
39	44.64355032	-63.59630844	16	11	12	11	2482	277	314	4357	3326
40	44.68678339	-63.48125213	28	23	13	13	4189	468	530	13316	6994
41	44.86189018	-63.59794484	41	39	26	26	3824	427	484	17570	12714
42	44.6363833	-63.57807438	18	13	13	13	4771	533	604	9453	7615
43	44.8235357	-63.74960601	32	34	28	28	5930	662	750	21192	20707
44	44.69733262	-63.54502517	24	19	6	6	3538	395	448	9399	2694
45	44.66039457	-63.55037771	21	16	10	10	3343	373	423	7934	4234
46	44.63705541	-63.70779133	6	12	24	12	4055	453	513	2660	6152
47	44.72645969	-63.51571121	30	25	10	10	3262	364	413	10902	4118
48	44.87314108	-63.68056058	39	40	31	31	8487	947	1074	36719	32783
49	44.69692684	-63.65770468	16	13	19	13	7362	822	931	13495	12446
50	44.82259609	-63.63064612	35	34	22	22	5372	600	680	21209	14848
51	44.66080316	-63.59374908	17	11	10	10	2562	286	324	4739	3327
52	44.71926465	-63.72399177	16	18	22	18	7185	802	909	12447	16296
53	44.64855884	-63.6209804	14	9	13	9	3544	396	448	5606	4144
54	44.57352041	-63.5640911	21	13	25	13	2196	245	278	5173	3719
55	44.6684083	-63.64111845	14	11	15	11	4863	543	615	7844	7053
56	44.67843358	-63.53412449	23	18	9	9	2015	225	255	5158	2202
57	44.77480064	-63.68637664	28	26	17	17	4069	454	515	12596	8929
58	44.75265663	-63.40431317	60	54	45	45	6508	726	823	43304	37193
59	44.67745286	-63.67207827	14	10	18	10	3358	375	425	5080	4458
60	44.59586001	-63.6152198	15	8	19	8	3868	432	489	6673	3789
61	44.68017983	-63.59804118	21	16	7	7	2014	225	255	4686	1804
62	44.71204839	-63.6755056	17	16	17	16	9580	1069	1212	18640	18929
63	44.73139852	-63.66938999	20	19	15	15	3634	406	460	8091	7019
64	44.7612874	-63.65250487	26	25	14	14	3313	370	419	9746	5992
65	44.71655067	-63.68737215	18	16	17	16	2801	313	354	5628	5736
66	44.70964213	-63.58181578	25	20	2	2	6569	733	831	18096	1599
67	44.693572	-63.61496549	23	18	6	6	1317	147	167	3417	1027
68	45.00477529	-62.57513063	155	155	139	139	3221	360	407	55665	56710
69	44.73878954	-63.8492522	27	31	34	31	5481	612	693	16618	21744
70	44.65084349	-63.64560952	12	9	15	9	7375	823	933	9789	8323

71	44.60959441	-63.58926946	17	9	19	9	6194	691	784	11672	7183
72	44.774451	-63.58453873	31	29	13	13	2142	239	271	7355	3536
73	44.63085965	-63.58847276	17	12	14	12	1808	202	229	3423	2706
74	44.48859548	-63.64031995	24	16	35	16	6430	718	813	17094	13098
75	44.6600863	-63.48493446	28	23	14	14	3606	403	456	11217	6191
76	44.765146	-63.69183225	27	25	18	18	6101	681	772	18084	13996
77	44.62829269	-63.61265973	16	9	15	9	4248	474	537	7370	4594
78	44.73489351	-63.23282781	55	50	41	41	5693	635	720	35183	29477
79	44.63782298	-63.59578051	16	11	13	11	3129	349	396	5543	4251
80	44.65932353	-63.64561589	13	10	15	10	4726	528	598	6614	5720
81	44.65578597	-63.73846944	7	11	23	11	5497	614	695	4156	7629
82	44.90940097	-63.39614355	55	56	40	40	4284	478	542	26509	21562
83	44.67838578	-63.48638429	28	23	13	13	3191	356	404	9891	5054
84	44.77555293	-63.65489642	28	27	16	16	3509	392	444	11095	7222
85	44.66617452	-63.36061158	39	34	24	24	6152	687	778	26871	18562
86	44.67845933	-63.51691218	25	20	10	10	3082	344	390	8739	4059
87	44.62224847	-63.62214602	15	7	16	7	4829	539	611	8020	4377
88	44.64832354	-63.59844379	16	11	11	11	5631	629	712	9933	7600
89	44.67276629	-63.61955325	16	13	13	13	5301	592	671	9572	8504
90	44.6638176	-63.51789491	25	19	11	11	8522	951	1078	23421	12032
91	44.65699167	-63.62083544	15	10	13	10	4594	513	581	7487	5915
92	44.63372875	-63.87581775	22	26	38	26	6568	733	831	15773	21364
Total	-	-	-	-	-	-	403131	45000	51000	1155315	903903

Table C-2 Summary of transportation inventory for sludge collection from each wastewater treatment facility to biosolid processing facility (HRM scenario) and Otter lake (for novel biorefinery scenario)

Facility name	Lat	Long	To BPF, km	To Otterlake, km	Treated water in 2017, m3	Normalized sludge	transport to BPF (HRM Scenario), t*km	transport to Otterlake (NB Scenario), t*km
Halifax	44.6536598	-63.57957625	37	18	52272321	17354	646812	304327
Dartmouth	44.65902023	-63.55544309	35	21	18818967	6248	215671	131844
Mill Cove	44.71654675	-63.67251581	29	19	8652553	2873	84204	53908
Eastern Passage	44.62519929	-63.51413583	39	26	5161571	1714	67639	45396
Aerotech	44.8567213	-63.52673406	8	42	304573	101	784	4229
Lakeside Timberlea	44.65067077	-63.73086068	43	6	897691	298	12871	1799
Herring Cove	44.56428037	-63.56082626	51	21	3633821	1206	61839	25597
Uplands Park	44.7261787	-63.73691427	35	17	30251	10	349	173

Springfield Lake	44.81476903	-63.73167705	34	30	209398	70	2395	2101
Frame Subdivision	44.77612983	-63.58345291	20	31	6616	2	45	68
North Preston	44.74855987	-63.46153652	38	35	244407	81	3072	2815
Lockview-MacPherson	44.82081024	-63.61610545	18	34	53819	18	330	611
Middle Musquodoboit	45.05323856	-63.12345584	43	87	71195	24	1015	2046
Wellingtone	44.85692201	-63.61668103	24	39	6752	2	54	88
Total	-	-			90363935	30000	1097079	575002

Appendix D: openLCA Results

Table D-1 Summary of each impact category value for different sub processes

Indicator	Fossil resource scarcity	Freshwater eutrophication	Land use	Marine eutrophication	Terrestrial acidification	Water consumption
Unit	kg oil eq	kg P eq	m^2a crop eq	kg N eq	kg SO_2 eq	m^3
HRM BPF	7.6E+05	-1.1E+02	-2.6E+04	-8.5E+00	5.2E+03	-1.5E+06
HRM land	1.0E+06	8.2E+03	6.6E+02	4.2E+04	3.4E+04	1.9E+07
NB Comp	3.2E+05	5.9E+02	-1.1E+04	7.3E+01	4.4E+04	5.7E+05
HRM Comp	7.0E+05	1.2E+03	-2.2E+04	1.4E+02	8.7E+04	1.2E+06
NB Land	7.3E+05	6.0E+03	1.4E+03	3.2E+04	2.3E+04	1.4E+07
NB AD	-9.2E+06	-1.7E+04	-9.8E+03	-1.3E+03	-2.2E+05	-3.0E+07
HRM, Transportation	1.2E+06	1.4E+02	3.9E+04	2.3E+01	1.2E+04	1.4E+06
NB, Transportation	1.2E+06	1.1E+02	2.1E+04	2.1E+01	1.2E+04	1.2E+06
HRM, Total	3.6E+06	9.4E+03	-8.4E+03	4.3E+04	1.4E+05	2.0E+07
NB, Total	-8.1E+06	-1.1E+04	-2.0E+04	3.1E+04	-1.5E+05	-1.5E+07

Table D-2 Break down results of each impact category for biorefinery product system

Case	Process	FRS	FE	GWP	Land Use	ME	TA	Water Con.
Baseline	Total	-2.67E+07	-4.28E+04	-9.89E+07	-1.15E+05	-3.27E+03	-6.37E+05	-1.11E+08
	Other	5.31E+05	5.73E+02	1.43E+06	3.08E+03	3.67E+02	1.09E+04	3.15E+06
	soybean	-9.91E+03	-1.19E+02	-6.51E+04	-5.94E+02	-8.71E+02	-3.68E+02	-9.21E+04
	Struvite P	-2.04E+05	-2.18E+02	-4.34E+05	-1.03E+05	-2.31E+01	-5.31E+03	-1.50E+06
	Struvite N	-4.01E+05	-4.29E+02	-8.51E+05	-2.03E+05	-4.54E+01	-1.04E+04	-2.94E+06
	Heat	-6.51E+05	-3.94E+01	-1.93E+06	-1.35E+02	-3.06E+00	-3.51E+03	-5.08E+05
	AmS	-6.91E+05	-9.55E+02	-2.32E+06	97.20117	-5.78E+01	-1.06E+04	-4.94E+06
	Electricity	-2.53E+07	-4.16E+04	-9.48E+07	1.89E+05	-2.64E+03	-6.18E+05	-1.04E+08
SENS1	Total	-1.61E+06	-1.39E+03	-5.18E+06	-3.07E+05	-6.53E+02	-4.85E+04	-4.15E+08
	Other	5.31E+05	5.73E+02	1.43E+06	3.08E+03	3.67E+02	1.09E+04	3.15E+06
	soybean	-9.91E+03	-1.19E+02	-6.51E+04	-5.94E+02	-8.71E+02	-3.68E+02	-9.21E+04
	Struvite P	-2.04E+05	-2.18E+02	-4.34E+05	-1.03E+05	-2.31E+01	-5.31E+03	-1.50E+06
	Struvite N	-4.01E+05	-4.29E+02	-8.51E+05	-2.03E+05	-4.54E+01	-1.04E+04	-2.94E+06
	Heat	-6.51E+05	-3.94E+01	-1.93E+06	-1.35E+02	-3.06E+00	-3.51E+03	-5.08E+05
	AmS	-6.91E+05	-9.55E+02	-2.32E+06	97.20129	-5.78E+01	-1.06E+04	-4.94E+06

	Electricity	-1.88E+05	-2.00E+02	-1.02E+06	-3.58E+03	-2.02E+01	-2.92E+04	-4.08E+08
SENS2	Total	-1.14E+07	-1.32E+04	-3.95E+07	-2.57E+05	-2.07E+03	-3.25E+05	-2.13E+08
	Other	5.31E+05	5.73E+02	1.43E+06	3.08E+03	3.67E+02	1.09E+04	3.15E+06
	soybean	-9.91E+03	-1.19E+02	-6.51E+04	-5.94E+02	-8.71E+02	-3.68E+02	-9.21E+04
	Struvite P	-2.04E+05	-2.18E+02	-4.34E+05	-1.03E+05	-2.31E+01	-5.31E+03	-1.50E+06
	Struvite N	-4.01E+05	-4.29E+02	-8.51E+05	-2.03E+05	-4.54E+01	-1.04E+04	-2.94E+06
	Heat	-6.51E+05	-3.94E+01	-1.93E+06	-1.35E+02	-3.06E+00	-3.51E+03	-5.08E+05
	AmS	-6.91E+05	-9.55E+02	-2.32E+06	-1.73E+02	-5.78E+01	-1.06E+04	-4.94E+06
	Electricity	-9.96E+06	-1.20E+04	-3.53E+07	4.67E+04	-1.43E+03	-3.06E+05	-2.07E+08