4-2016

Sustainable Management of Food Supply-Chain Resources in New York State

Jacqueline H. Ebner
jhe5003@rit.edu

SUSTAINABLE MANAGEMENT OF FOOD SUPPLY-CHAIN RESOURCES IN NEW YORK STATE

By

Jacqueline H. Ebner

A Dissertation
Submitted in Partial Fulfillment
of the Requirements for the Degree of
Doctor of Philosophy
In
Sustainability

Department of Sustainability
Golisano Institute for Sustainability
Rochester Institute of Technology

April 2016

Author: ____
Sustainability Program

Certified by: ____
Dr. Thomas A. Trabold
Associate Professor and Department Head, Sustainability Program

Certified by: ____
Dr. Nabil Nasr
Assistant Provost and Director, Golisano Institute for Sustainability and CIMS

Notice

Funding provided by the New York State Pollution Prevention Institute through a grant from the New York State Department of Environmental Conservation.

Any opinions, results, findings and/or interpretations of data contained herein are the responsibility of author(s) and do not necessarily represent the opinions, interpretations or policy of the State.

Sustainable Management of Food Supply Chain Resources in New York State

by

Jacqueline H. Ebner

Submitted by Jacqueline H. Ebner in partial fulfillment of the requirements for the degree of Doctor of Philosophy in Sustainability and accepted on behalf of the Rochester Institute of Technology by the dissertation committee.

We, the undersigned members of the Faculty of the Rochester Institute of Technology, certify that we have advised and/or supervised the candidate on the work described in this dissertation. We further certify that we have reviewed the dissertation manuscript and approve it in partial fulfillment of the requirements of the degree of Doctor of Philosophy in Sustainability.

Approved by:

Dr. Jeffrey S. Lodge ______________________________
(Outside Examiner and Chair) Date

Dr. Thomas A. Trabold ______________________________
(Dissertation Advisor)

Dr. Anahita A. Williamson ______________________________

Dr. Gabrielle Gaustad ______________________________

Dr. Rodrigo A. Labatut ______________________________
(External Examiner)

Acknowledgements

I would like to thank the many who have contributed to this effort. First, I would like to thank my dissertation committee beginning with my advisor Dr. Thomas Trabold for his steady and wise guidance throughout my research. I also thank Dr. Anahita Williamson who was my primary advisor for two years for her encouragement and support and for providing editorial comments as a member of my dissertation committee. Finally, I thank my advisor Dr. Gabrielle Gaustad for her council and interest in my research. I am also grateful to Dr. Rodrigo Labatut of Cornell University for his willingness to share his knowledge and collaborate with me and from whom I have learned a great deal. Also, I would like to acknowledge the extended GIS faculty members Dr. Jeffrey Lodge and Dr. Brian Tomaszewski for their advice and willingness to share their expertise. Finally, Dr. Fritz Ebner, my husband, who has on occasion been my advisor, collaborator and proof-reader but most of all whose support and love has made this all possible. I dedicate this work to him, my children and the friends and family who have supported me. I also gratefully acknowledge the New York State Pollution Prevention Institute (NYSP2I) which provided my Graduate Research Assistantship through a grant from the NYS Department of Environmental Conservation.

Abstract

A sustainable food supply chain (FSC) is at the nexus of several critical global challenges including hunger, resource scarcity, climate change, poverty, energy security and economic growth. However, managing FSC resources in a sustainable manner is complex and data to support this goal is lacking. This dissertation addressed four knowledge gaps by applying a variety of analytical and experimental tools to the New York State FSC.

First, a cradle-to-grave analysis of the New York State FSC was conducted. Resources leaving the FSC from primary production (post-harvest) through to consumption were defined and characterized. Surveys and literature were used to estimate FSC resources and factors were provided for several sectors and sub-sectors including the Educational sector. Material flows through the utilization pathways in New York State were analyzed. It was estimated that over 3.5 million t/yr of solid resources were generated. Resource utilization pathways including donation were estimated to treat approximately 6% of these resources. An additional 22 million m^3/yr of low solid resources primarily from the food processors was also estimated and analyzed.

In the next chapter, climate change impacts of utilization pathways emerging in the State were analyzed. Two comprehensive lifecycle assessments (LCAs) were conducted to assess climate change impacts. The first was based upon primary data collected from the largest on-farm anaerobic digester in the State, which co-digests dairy manure and industrial food wastes. The results showed a net negative climate change of 37.5 kg CO_2e/t influent processed when compared to the reference case. Displacement of grid electricity provided the largest reduction, followed by avoidance of alternative

food waste disposal options and reduced impacts associated with storage of digestate vs. undigested manure. Sensitivity analysis showed that using feedstock diverted from high impact disposal pathways, control of digester emissions, and managing digestate storage emissions were opportunities to improve climate change benefits. The second LCA was based upon a small-scale, distributed waste-to-ethanol process. This analysis was based upon data from an operating pilot plant facility, co-fermenting industrial and retail FSC resources. The climate change impacts for the processing phase were estimated to be comparable to those associated commercial ethanol production, however when considering the avoidance waste disposal for FSC resources used as feedstock, the result was a net negative impact of 338 kg CO_2e/MJ fuel produced.

The following chapter evaluated the potential of several significant New York State FSC resources as feedstock for biogas production. Twenty-four source-separated, commercial substrates from the retail and food processing sector were characterized and tested in bench-scale bio-methane potential (BMP) tests. Substrates were also combined with dairy manure and other substrates to assess synergistic or antagonistic effects associated with co-digestion. Key bio-methane kinetic parameters including bio-methane potential, apparent hydrolysis rate constant and co-digestion indices were reported. Substrates with high fat content demonstrated higher potential for bio-methane generation. Substrates rich in readily hydrolysable carbohydrates and fats showed more complete bio-degradation. Measured bio-methane potential was the product of both of these factors. Bio-methane production of co-digested substrates was close to that of the weighted average of the individual substrates with a slight synergistic bias (-5%/+20% on average). However, co-digestion generally resulted in an increase in apparent hydrolysis rate relative to that predicted by the combination of individual substrates.

Finally, the impact of FSC resource characteristics on greenhouse gas (GHG) emissions related to utilization of those resources analyzed. An open source model (ORCAS) was developed to assess the climate change impacts of several NYS resource utilization pathways. Data gathered in the previous chapters were used to select the most relevant FSC resources and provide characterization data, which was used to calculate the impacts of these resources across the different utilization pathways. These results were compared to the generally reported results based upon the characteristics of municipal solid waste food scraps (MSWFW). The comparison showed that resource characteristics can have a significant impact on net GHG emissions, most notably in the case of landfilling. Linear formulae were also provided to estimate impacts based upon key resource parameters. A Monte Carlo simulation was performed and model uncertainty was discussed.

Table of Contents

List of Tables

List of Figures

Abbreviations

ABV	Alcohol by volume (%)
AD	Anaerobic digestion
AcoD	Anaerobic co-digestion
B_o	Bio-methane potential
B_u	Theoretical bio-methane potential
BMP	Bio-methane potential assay
BOD	Biochemical oxygen demand (mg/kg)
BUD	Beneficial use determination
C	Carbon
CL	Crude Lipids (%)
CP	Crude Protein (%)
CF	Capacity factor
CS	Carbon storage
COD	Chemical oxygen demand (mg/L)
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalents
CH_4	Methane
DAF	Dissolved air floatation waste
DDGS	Dried distillers grains and solubles
EtOH	Ethanol
FOIL	Freedom of information act
FFS	Feed fuel slurry
FSC	Food supply chain

GHG	Greenhouse gasses
GIS	Geographic information system
GTW	Grease trap waste
HRT	Hydraulic retention time
IFW	Industrial food waste
K	Potassium
kg	Kilogram
kWh	Kilowatt hour
LFG	Landfill gas
m^3	Cubic meters
MCF	Methane conversion factor
N	Nitrogen
NAICS	North American Industry Classification System
N_2	Nitrogen gas
NO	Nitric Oxide
N_2O	Nitrous oxide
NH_3	Ammonia
NYSDEC	New York Department of Environmental Conservation
NYS	New York State
OLR	Organic loading rate (kg VS/ m^3-d)
ORL	Organic resource locator
ORCAS	Organic resource climate impact assessment simulator
P	Phosphorous
t	Tons (metric tons)
TAN	Total ammonaical-N (mg/kg)

TKN	Total Kjeldahl Nitrogen (mg/kg)
TPEY	Tons per employee year (short tons)
TS	Total solids (%)
TSS	Total suspended solids (mg/kg)
SSF	Simultaneous saccharification and fermentation
USDA	United States Department of Agricultural
VS	Volatile solids
ww	Wastewater
WWT	Wastewater treatment

Chapter 1 Introduction

1.1. *Background*

A sustainable food supply chain (FSC) is at the nexus of several critical global challenges (Fig 1-1). The World Resource Report identifies reducing food loss and waste as one of the solutions to what they term the "great balancing act" of feeding more than 9 billion people by 2050 in a manner that advances social and economic development while reducing pressure on ecosystems, climate and water resources (Lipinski et al., 2013). Inefficiencies in the FSC, resulting in losses and waste, reduce food availability and also consume energy, water and other resources. Precise estimates of resources leaving the FSC are illusive, however the Food and Agriculture Organization of the United Nations (FAO) estimates that 32 percent of all food produced in the world is lost or wasted (Lipinski et al., 2013). Food loss and waste is estimated at 133 billion pounds annually in the U.S. (Buzby et al., 2014) and both reduces the food supply and consumes energy, water and other resources. This quantity is based only on edible food mass leading to human consumption and thus does not include by-products or inedible scrap, or food grown for feed or bioenergy, which taken as a whole represents a tremendous source of renewable resources.

Recycling or up-cycling FSC resources can provide nutrients, chemicals, fuels or other high value products. When converted to bio-fuels, FSC resources contribute to energy independence and reduce the climate change impacts associated with fossil fuel use without posing a conflict with food production or land use. Thus several states have included various waste-to-energy (WtE) fuels in the list of qualifying renewables presented in their Renewable Portfolio Standards (RPS) (U.S. EPA, 2015). Disposal of FSC resources also often comes with significant economic and environmental costs. According to the U.S. EPA, the nation spends about $1 billion a year to dispose of food waste (U.S. EPA/USDA, 2015). Environmental impacts, such as the release of greenhouse gases (GHG) resulting from treatment of FSC resources and concerns over land use have resulted in increasing regulation of landfill disposal in parts of Europe and the U.S. Recycling of food supply waste can return valuable nutrients to the ailing soils. Thus management of the FSC resources is clearly one of the great sustainability frontiers addressing critical social, economic and environmental goals.

Figure 1-1: The sustainable food supply chain intersects several global challenges including hunger, resource scarcity, climate change, poverty, energy security and economic growth.

However, despite its importance, management of FSC resources is difficult and has historically received little concerted attention resulting in significant knowledge gaps. The objective of this work is to address some of these gaps through application of a variety of analytical and experimental tools applied to New York FSC resources.

Food systems are at the core of the New York State economy. Approximately one-fourth of the State's land is devoted to agriculture (OSC, 2015). The food processing industry is estimated to generate over $19 Billion dollars in annual revenues and to employ over 54,000 (US Census Bureau, 2007). Simultaneous alternative treatments to landfilling of organic wastes are being actively pursued as a way to mitigate climate change impacts associated with methane production and to reduce land use conflicts (Massachusetts, 2013). While alternative utilization pathways are available, several compelling questions should be answered in order to informed policy to guide this transition. This work specifically seeks to address the following questions:

1. What FSC resources are generated in New York State and how are they currently utilized?
2. What are the net greenhouse gas emissions reductions achievable with anaerobic co-digestion (AcoD) and waste-to-ethanol systems?
3. How can available commercial food waste resources be combined to maximize bio-methane production in AcoD systems?
4. How do the specific resource characteristics influence our choice of "best" alternative pathway?

1.2. *Dissertation structure*

This work consists of four major research segments each comprising a chapter of this dissertation as follows:

Chapter 2: Analysis of New York State FSC resources: Includes a framework for data collection and analysis of available data

Chapter 3: Climate change impacts of emerging food supply chain utilization pathways: Consists of lifecycle greenhouse gas (GHG) analyses based on primary data from two NYS facilities

Chapter 4: Evaluation of anaerobic digestion of commercial food waste as: characterizing biochemical parameters and synergistic effects: Experimental work to provide a data set related to anaerobic digestion of one of the State's fastest growing utilization pathways.

Chapter 5: Comparison of climate change impacts for treatment of specific FSC resources: Combines the experimental data (Chapter 4) to extends the site based lifecycle assessments (Chapter 3) to generalized models for assessing FSC resource specific climate change impacts for the State's most common utilization pathways

Chapter 2 Analysis of New York State FSC resources

Management of FSC resources has gained increasing attention globally, mentioned in 2 of the 17 goals of the UN's 2030 Agenda for Sustainable Development (UN News Centre, 2015). Also, in the U.S. the USDA has recently announced of the first-ever national food waste reduction goal, calling for a 50-percent reduction by 2030 (Tagtow et al., 2015). However, after decades of sporadic effort, data on FSC resources is scant and tailored to varying objectives (Parfitt et al., 2010). This chapter expands this body of knowledge by conducting an analysis of the FSC of New York State. The Introduction provides the framework for analysis, including key terms and definitions followed by a history of food waste analysis in the literature. The Methods section discusses the data collection process and the development of FSC resource generation factors. Three main analyses are described: quantification of FSC resources, geographic analysis of FSC resources and utilization pathways, and material flow analysis of FSC resources. Study limitations, gaps and future work are discussed in the Results.

2.1. *Introduction*

2.1.1 Framework and definitions

Because management of the FSC intersects many different goals (e.g., hunger elimination, climate change mitigation, economic development, etc.) studies to evaluate the FSC have had varied approaches and objectives. Therefore a foundational step is to define key terms and provide a framework and objectives for analysis.

Technosphere is the "man-made" environment, that which is modified by humans, for use in human activities. Supply chains are subsystems of the *technosphere* that convert natural resources from the *ecosphere* into products that are used to deliver services to humans (DeWulf et al., 2016)

Food supply chain (FSC) is defined as the system of interacting processes that produce food for human consumption. This is sometimes termed the "farm to fork". In this analysis, the system is constrained to begin post-harvest (at the farm gate) and to continue through the steps of processing, distribution/retail and consumption. (Fig 2-1)

Food loss represents the edible amount of post-harvest food that is available for human consumption but not consumed for any reason. It includes cooking loss and natural shrinkage (e.g. moisture loss); loss from mold, pests, or inadequate climate control and food waste (Gustavsson et al., 2011; Kirkendall, 2015)

Food waste is a subset of food loss and occurs when an edible item goes unconsumed, as in food discarded by retailers due to color or appearance and plate waste by consumers. Thus food waste occurs only at the retail and consumption stages (Gustavsson et al., 2011; Kirkendall, 2015)

Food supply chain (FSC) resources are secondary resources which consist of whole and/or parts of food which enter the FSC and do not pass through the entire food chain, following the approach proposed by Östergren et al. (2014) and Soethoudt and Timmermans (2013). Note that food waste and food loss are measured only for products that are intended for human consumption, and thus exclude parts or products which are non-edible, while the definition of FSC resources does not.

Utilization pathways are processes and technologies used to treat FSC resources.

Utilization pathways receive FSC resources as inputs (either free of charge, as a source of revenue, or at a cost) and manage the resource until it is either returned to the ecosphere (soil, water or air) or the technosphere (the FSC or another supply chain or consumer market).

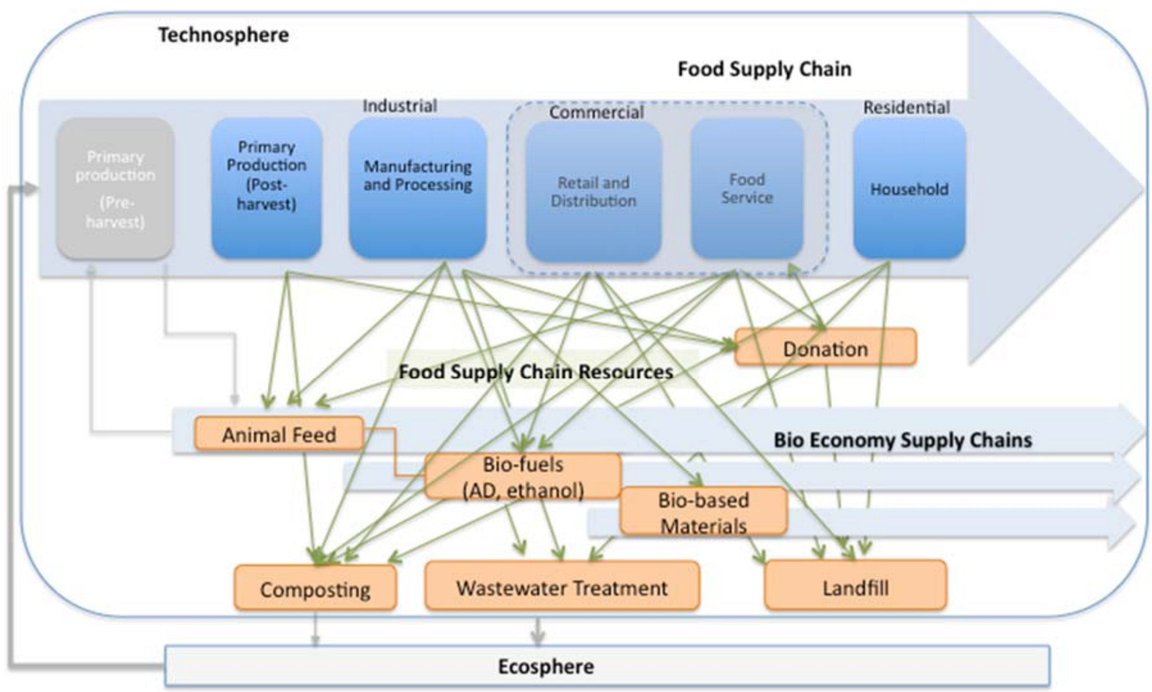


Figure 2-1: Food production occurs along the Food Supply Chain (FSC). FSC resources exit the food supply chain to a utilization pathway (shown in orange). Utilization pathways recycle resources within the Technosphere or return them to the Ecosphere. Resources are recycled into the Technosphere via the food, bio-economy or another supply chain. Resources recycled to the Ecosphere are again available to the FSC or Bio-Economy supply chains. (Modified from Fusions Definitional Framework (Ostergren et al., 2014)).

FSC resources are generated at every level of the FSC. They can take the form of organic matter in high strength wastewater, by-products of production processes, scrap or non-edible portions and discarded food.

2.1.2 History of FSC analysis

In 1945 when the Food and Agriculture Organization of the United Nations (FAO) was established, reduction of food loss was part of its mandate (Parfitt et al., 2010). At the VIIth special session of the United Nations in 1975, then U.S. Secretary of State Henry Kissinger, realizing the link between FSC management and global hunger, strongly recommended a resolution to cut post-harvest food loss 50% by 1985. (Hongladarom, 2015). The resolution was adopted in 1975 and a 1976 report concluded that lack of information, along with lack of infrastructure and investment, were barriers to reducing food loss in the supply chain[1]. While some early progress was made relating to one or two cereal crops in developing countries, little more was reported on progress toward this original goal. Within the past decade the call has once again gone out to half food losses and wastage, this time by 2050 (Lundqvist et al., 2008; Tagtow et al., 2015). While the problems of poor data quality, complexity in FSCs and different definitions remain barriers, actions over the past several years signal growing momentum to tackle the problem.

[1] Food and Agriculture Organization (FAO) of the UN, "Launch of the G20 Technical Platform on the Measurement and Reduction of Food Loss and Waste", May, 2015, http://www.fao.org/about/meetings/council/cl153/side-events/technical-platform/en/

At the 2015 United Nations Climate Change Conference, (COP 21 Paris), the Food and Agricultural Organization (FAO) and the International Food Policy Research Institute announced the G20 Technical Platform on the Measurement and Reduction of Food Loss and Waste (FAO, 2015). Although the U.S. lags behind the European Union, United Kingdom and Denmark, efforts in the U.S. are gaining momentum. National progress has developed out of efforts to track food supply and national diet and nutritional patterns. In the mid 1990's the USDA's ERS (Economic Resource Service) expanded the Food Availability Data Series (FADS) to track per capita daily intake. The loss adjusted food availability (LAFA) series was created by subtracting losses such as spoilage and plate waste from commodity production, import and export data. Loss estimation coefficients were taken from published reports or discussions with commodity experts (most dated in the mid-1970s or earlier). From this effort a report was issued highlighting the magnitude of losses of edible food at the retail, food service and consumer levels and seeking solutions to reduce losses through recovery, recycling and education (Kantor et al., 1997). In 2005 the ERS recognized the need to systematically update and improve all loss assumptions for each commodity. The years that followed have seen efforts to improve loss estimates for several commodities and at the primary, retail and consumer levels. Today, while it is still acknowledged that data quality can be improved, the FADS and LAFA series track FSC losses for several hundred commodities[2]. Although this data series cannot be used to estimate FSC resources from individual generators, it can provide information on the overall composition of these FSC resources and losses at each level of the FSC.

Efforts at the State level have followed a different path, largely motivated by waste diversion or renewable energy goals. These efforts have typically included data and geographic information to assist in development of organics diversion infrastructure. Unlike the top-down approach at the Federal level, they usually apply a bottoms-up methodology using waste generation factors rather than loss factors. The waste generation factors are applied to some representative metric (e.g., numbers of employees, number of students, etc.) to estimate establishment or sector level FSC resources generated. The main focus of these studies has been on municipal solid waste (MSW) and the commercial and residential sectors. However, in several studies, waste generation factors were poorly documented and when traced back relate to studies conducted in the 1980s or 1990s (CDEP, 2001; Ma, 2006; MDEP, 2002; NCDENR, 2012). The state of California has conducted several statewide municipal solid waste (MSW) characterization studies wherein waste volumes were estimated based on waste audits conducted at several types of establishments throughout the State and characterization of the audited waste into categories including "food waste"[3](Calrecycles, 1999; 2006; 2009; 2014). These studies estimate the quantity of waste generated and in

---

[2] USDA, Loss Adjusted Availability data (LAFA), 2015c, http://www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system/.aspx#26705

[3] The term food waste here does not refer to the definitions used globally or at the national level, but rather to municipal solid waste food waste (MSWFW) or solid FSCR.

some reports the quantity disposed (via landfill, incineration or wastewater treatment) or diverted (recycled). Beyond MSW, a Michigan study (Safferman et al., 2007) motivated by water quality concerns estimated wastewater generated from fruit and vegetable processing by applying typical data on wastewater volumes and strength (TSS and BOD) to fruits and vegetables processed in the state. Ma (2006) used FSC resource generation factors and also surveyed several food processors in New York State in order to estimate statewide resources available for energy conversion. In 2007, Matteson and Jenkins (2007) performed a similar and more comprehensive assessment to quantify resources available for energy conversion in California.

The present research, while building on many of the efforts outlined above, differs in its broad holistic approach. It is not commodity-based nor restricted to the edible FSC like the national studies, but uses that data to provide information on composition at various stages. It expands on the methodologies used in many of the state studies by beginning work to quantify FSC resources for NYS. In doing so, a thorough review of the literature was conducted along with some primary data collection to assess and select FSC resource generation factors. In addition to characterizing FSC resources, data was also collected on utilization pathways. This data was then analyzed quantitatively, geographically and using a material flow analysis. The specific outcomes were as follows:

1. Provide a framework for analyzing New York State FSC resources.
2. Quantify New York State FSC resources and provide FSC resource generation factors.
3. Map FSC resource and utilization pathways to support market development.

4. Apply material flow analysis to identify trends, opportunities and challenges related to emerging FSC utilization technologies in New York State.
5. Identify knowledge gaps to inform technology development described in the remaining chapters of this dissertation and other work leading to the goal of a sustainable food supply chain for New York State.

This work is intended to inform planners, developers, municipalities and individual establishments in achieving social, environmental and economic goals for FSC resource management. This chapter should be viewed as a starting point for NYS. Available data and methodologies are thoroughly discussed to provide a foundation for other studies. Data gaps and suggestions to fill these gaps are also discussed.

2.2. *Methods*

2.2.1 Generation of FSC resources

A bottoms-up approach was taken to assess resources at each step of the FSC. Public and private databases, and data obtained through freedom of information law (FOIL) requests were used to identify New York State FSC resource generators along with significant characteristics. An initial focus was placed on larger generators.

FSC resource generation was estimated in some cases by applying a FSC resources generation factor. Interviews, surveys or primary data were also applied to supplement other data.

2.2.1.1 Primary production

Only post-harvest losses were considered in this analysis. Therefore crop residues and un-harvested crops were excluded.[4] Similarly, livestock production generates vast amounts of manure in the State, while this is not within the boundaries of this analysis some information is provided as reference.[5]

Therefore, FSC resources at the primary production level mainly consist of post-harvest perishable crop losses[6].

Data from the USDA Agricultural Census for NY was used to identify the top crops for the state[7]. Loss factors from the USDA's LAFA database were then applied to estimate the weight of crops harvested but not sold[8].

2.2.1.2 Food manufacturing and processing

A query of the business database ReferenceUSA® (Infogroup, 2014) was used to identify and locate food manufacturers and processors in the NYS food supply chain

---

[4] Gunders et al. (2012) estimated that approximately 7% (but up to 50%) of crops planted are not harvested in the US. Reasons include pests, disease, weather, labor shortages, consumer quality standards and economics.

[5] The reader is directed to other work by the author for details on quantifying this resource (Chan et al., 2013; Ebner et al., 2014).

[6] Livestock morbidity losses from farm to retail were excluded at this time as they were assumed to be small.

[7] USDA National Agricultural Statistics Service (NASS), NYS Agricultural Overview 2014, http://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=NEW%20YORK

[8] USDA, Loss Adjusted Availability data (LAFA), 2015c, http://www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system/.aspx#26705

based upon North American Industry Classification System (NAICS) codes and criteria for number of employees and sales:

- NAICS 311-312 (all NAICS) [9]
- State of New York
- 5+ employees
- $1M+ in sales

The data was then reviewed and compared to industry databases or other sources to remove duplicates and improve accuracy. Food Processors were grouped into broad categories based upon the type of FSC resources generated (Table 2-1).

Table 2 -1: Food manufacturers and processor categories and descriptions

Category	**Description**
Bakery/Mill	Commercial bakeries, cookies, crackers, pasta, dough, flour mills, snacks and cereal manufacturers
Beverages/ Syrups/ Sauces	Makers of soft drinks, juices, sauces, dressings, flavorings, ciders
Breweries	Beer makers
Canning	Fruit, vegetable and specialty canning, jellies, including tomato sauce and apple sauce
Coffee/Tea/Tobacco	Coffee, tea and tobacco producers
Confectionary/ Candy	Candy makers, confectioners and sugar processors
Wineries	Wine makers
Dairy	Cheese, milk, yogurt, ice cream and butter creameries
Distillery	Maker of distilled spirits
Frozen foods	Frozen fruit, vegetables, meal and specialty item producers
Meat /Seafood	Slaughter houses, commercial butchers, meat packers, hatcheries
Spice/ Dehydrated	Spice manufacturer and dehydrated foods
Misc.	Nut butters, soup, gourmet food, soy products, rendering, other

[9] Facilities bottling water or manufacturing ice were excluded.

A variety of techniques have been employed in previous studies to estimate resource generation at the processor level and have generally concluded that estimation is tedious and challenging (Amon et al., 2012; Ma, 2006; Safferman et al., 2007). Generalized formulae are difficult to apply to food processors for several reasons. Differences in final products within a category (e.g., meat packer vs. meat curing house) can generate different FSC resource profiles as will different manufacturing or waste treatment processes. Also, technology advances and process improvements also make FSC resource dynamic and estimates quickly obsolete. Moreover, limited data is publicly available and many processors are reluctant to share data on FSC resources either out of proprietary concerns or concerns over drawing unwanted attention from regulators.

This study therefore used two main techniques to obtain data on the food manufacturing and processing sector: 1) survey of food processors, and 2) publicly available data obtained through freedom of information act law (FOIL) requests or reports. This data was used to presents a broad representation of FSC management at this level. The analysis is viewed as a starting point for further analysis and discussion.

Survey of food processors: As part of his dissertation on a spatial decision support system for organic waste in New York, Ma (2006) collected data from 33 food processors in New York State. However, given the pace of change in the market and technology of the food processing industry, it was determined that additional data should be obtained as part of the current research program. A phone survey was prepared in 2013 and several food processors throughout the state were contacted however a very poor response was initially achieved. A second survey was attempted with a focus on companies with which an existing relationship had already been established. In the latter case, the survey was administered online through email distribution, by phone and/or in person. Respondents were asked to provide information on their company and the volume and characteristics of FSC resources leaving their plant in the form of wastewater or solid waste. In some cases, information on waste treatment on-site was also provided. The survey form is provided in Appendix A.

Public record: In some communities, companies that utilize the publicly owned wastewater treatment works (POTW) are required to pay a surcharge for discharges that have high total suspended solids (TSS) and/or biological oxygen demand (BOD), or other characteristics (e.g., high phosphorous or chlorine content). These are classified as "high strength" wastewater discharges. A FOIL request was made to all of the counties in NYS requesting this data. The response was limited because not all counties operate their POTW, maintain records of high strength discharges or charge a surcharge. The largest source of data was obtained from Monroe County, where RIT is located.

In addition the New York State Department of Conservation (NYS DEC) prepares reports on a variety of other activities related to FSC resource utilization. This included data on resources that are treated at a registered organics recycling facility and regulated resources that are land applied or diverted to feed animals or to another beneficial use. These reports are discussed in the in the waste utilization pathway section below.

2.2.1.3 Retail and distribution

The retail/distribution sector consists of markets, wholesalers and distribution centers. In an effort to focus on larger generators the initial focus was on supermarkets, convenience stores and big box stores with grocery sections. A marketing database query (Infogroup, 2015) was made as follows:

- Primary NAICS keywords “supermarkets, convenience stores and grocery stores”
- Also Walmart and Target stores with 445110 in all NAICS
- State of NY
- 5+ employees
- $1M+ in sales

Data from the California waste characterization studies was used to develop a FSC resource generation factor based upon number of employees. This factor compared well with data collected for 6 NY supermarket stores that tracked data on FSC resources diverted utilization program for one year (2012-13). However this factor was higher than one based upon studies from the 1990s possibly due to the expanded food preparation operations at many modern supermarkets (Table 2-2).

While averages agreed significant variability between stores was observed. This is presumed to be due to different store operations and thus when seeking a factor to estimate store level FSC resources, considerations such as the amount of produce and prepared foods on-site should be used to adjust the resource generation factor accordingly.

The only data available for big box retail stores (ie. Wal-Mart and Target) was from the California study (Calrecyles, 2006). No data specific to convenience stores could be found so the supermarket factor was applied.

Table 2-2: Literature review of Supermarket FSC resource generation factors

Source	Description	kg/employee-yr
(Calrecyle, 2006)	food stores	2,104
(Calrecyle, 2014)	food and beverage stores	1,835
NY Grocery chain (2014)	store 1	4,355
	store 2	697
	store 3	2,236
	store 4	660
	store 5	1,427
	store 6	2,591
	average	1,994
(CDEP, 2001) Literature review	Kings County, 1995 (survey)	1,300
	King County, 1995 (audit)	1,482
	Newell et al., 1993	1,291
	Jacob, 1993 (20,000 sf stores)	1,573
	Jacob, 1993 (30,000 sf stores)	1,309
	Jacob, 1993 (45,000 sf stores)	1,227
	Newell and Snyder, 1996	1,327
	Grocery Industry committee, 1991a	1,409
	Grocery and Industry Committee, 1991a	1,245
	Average	1,355
	Used in this study	**2000[b]**

[a] Converted from lbs/$1000

[b] rounded to nearest significant figure to indicate implied precision of the estimate

2.2.1.4 Food service and consumption

Food service and consumption was broken into 3 broad sectors and then several sub-sectors where feasible.

Institutions

There are a variety of institutions that generate FSC resources through food service and housing operations. Three sub-sectors of institutions were analyzed: education, health and medical and entertainment, lodging and restaurants.

Education: Kindergarten through grade 12 (K-12) schools were analyzed on a district level basis. Student enrollment data for public and private K-12 schools was collected from the NY State Education Department (NYSED) Information and Reporting Service (IRS)[10].

Studies that have estimated K-12 food supply chain resources (Griffin et al., Ma, 2006) have generally based their analyses on data from the late 1990s (Block, 2000, Hollingsworth et al., 1995). A thorough review of available literature was conducted to determine an appropriate FSC resource generation factor (Table 2-3). This included several more recent studies as well as data publicly reported by the Vermont Central school district compost program (Appendix A, Table A-1)[11]. The Vermont data was considered the most recent, extensive and relevant dataset.

For simplicity a single K-12 factor was used in this study, however, it has consistently been observed that greater resources are generated at the Elementary level with decreasing rates at middle and high school levels (Appendix A, Table A-2).

---

[10] NYS Education Department (NYSED) Information and Reporting Service (IRS), Public and Charter School Enrollment 1993-94 to 2013-14, accessed March 10, 2014, http://www.p12.nysed.gov/irs/statistics/enroll-n-staff/home.html

[11] It is acknowledged that is factor is actually a FSC resource diversion factor and not a true generation factor however is taken as reasonable in the context of the other factors (ie. it is higher than some generation studies.)

Therefore, this factor should be adjusted appropriately if used to estimate resources for an individual school. Furthermore, this analysis does not include liquid resources, however a study in Florida estimated milk waste to be nearly half the weight of solid resources generated at the Elementary level and thus may be an important resource to consider in some cases (Appendix A Table A-2).

Table 2-3: Literature review of FSC resource generation in K-12 schools

	kg/student-yr	Notes
Hollingsworth et al., 1992	23	6 Louisiana schools
Hollingsworth et al., 1995	31	7 Louisiana schools
Block, 2000	27	15 Kansas schools
MPCA, 2010	10	6 Minnesota schools
Wilke et al, 2014	9	3 Florida schools
Vermont, 2015	15	27 Vermont schools
Cascadia, 2014	11	51 CA educational facilities[a]
Used in this study	15	

[a] Includes all educational facilities based upon NAICS code, not just K-12

Information on NY State colleges and universities, including full year enrollment was obtained from the NYS Department of Education (NSED) research and information system (ORIS).[12]

The most commonly cited formula to estimate FSC resources for colleges and universities level is based upon a review of literature from 1997-2001 (CDEP, 2001).

[12] NYS Education Department (NYSED) Office and Reporting and Information Service (ORIS), Enrollment report, all schools, http://eservices.nysed.gov/orisre/mainservlet 2012 accessed Jan 2014.

Similar to the K-12 studies it used meal audit data from studies in the literature to arrive at a weight/meal estimate, which was multiplied by annual meals served at the institution per enrolled student, which was based upon a limited sample of expert estimates.

A thorough review was also conducted to determine an appropriate FSC resources generation factors for colleges and universities. It included peer-reviewed studies in the literature as well as publicly available data and reports from colleges and universities that conducted waste audits or employ organic waste diversion programs. Meal audit data from 11 institutions and campus level data from 13 institutions were analyzed and reported in Ebner et al. (2014). A summary of the results is included in Appendix A (Table A-3, Table A-4, Table A-5).

The results of this analysis showed that both the commonly used meal audit factor and the meals per enrolled student estimate may need to be revised. Furthermore, data on meals served at the institution per enrolled student is difficult to obtain. Therefore, a factor based upon institution level data was recommended. This factor was arrived at via a regression of the establishment level data collected (Fig 2- 2).

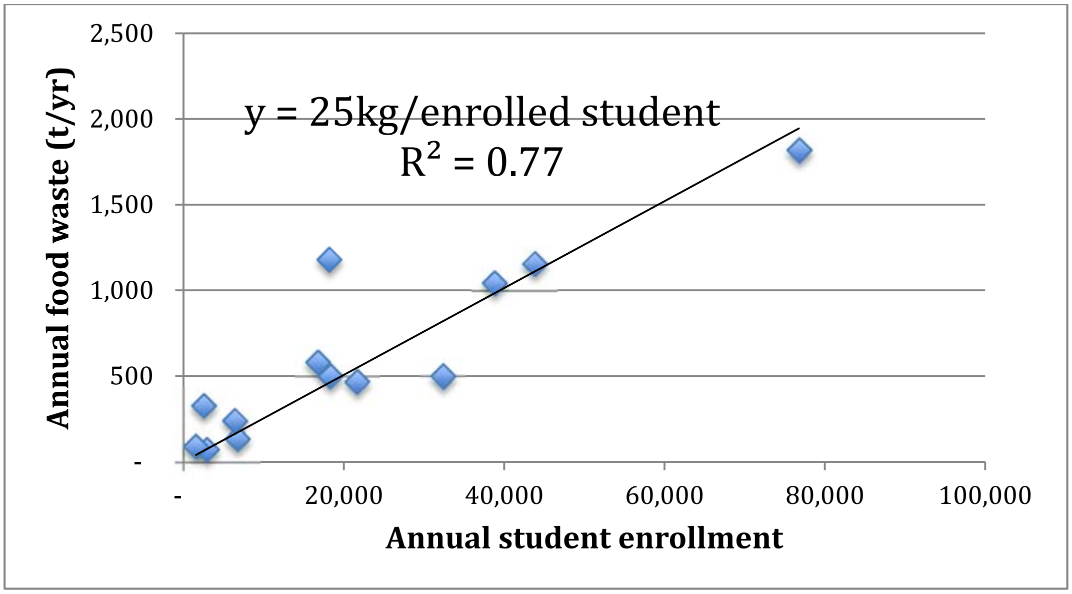


Figure 2-2: Regression of institution level food waste vs. annual student enrollment for colleges and universities

Reasons for poor fit of data points to the regression were attributed to: 1) institutions with higher or lower staff/faculty to enrolled student ratios (i.e., very small or very big schools); 2) schools with high rates of visitors to campus (i.e., big sports programs or research institutes); or 3) schools with very high or low access to off-campus food sources (ie., rural or urban). Thus adjustments in the FSC resource generation factor should be made when estimating resources for specific institutions that fall within these categories.

No data could be found to estimate FSC resources from community colleges, however a factor based upon 4-year residential schools was assumed not to apply. Expert interviews in the early Connecticut study suggested that community colleges serve approximately 1/4th as many meals as residential institutions (CDEP, 2001). In the absence of more recent data this estimation is applied to 2-year schools and the other factor used for 4-year schools.

Finally, low solids waste was not included at this time as it was shown to be a small contribution to FSC resources based upon cafeteria audits (Appendix Table A-3).

Health and Medical: Data from the NY Department of health including bed counts was obtained for nursing homes[13] and hospitals[14] in the State.

The FSC resource generation factor most commonly cited for hospitals and nursing homes can be traced to the CDEP study (2001). Similar to the factor for the college and university sub-sector, it used a waste per meal value based upon reviewed studies dating from the mid-1990s. This was then extrapolated to the institution level by multiplying by the number of meals served at the institution per bed, which was estimated by surveying 7 Connecticut health care institutions. This was compared to most recent data on this sector from the California audits which resulted in 1/5th the factor (Appendix A, Table A-7) (Calrecycle, 2014). The California factor was applied in the current research with improved data on hospital and nursing home FSC resource generation factors identified as an area for future work.

Government facilities: Data on correctional facilities including inmate counts for county jails and state and federal prisons was obtained through a FOIL request to the NY Department of Corrections and Community Supervision (NYDOCCS).

---

[13] NY State Department of Health (DOH), Adult Care Facility Annual Bed Census Data:, http://www.health.ny.gov/facilities/adult_care/

[14] NY State Department of Health (DOH), New York Hospitals by County, http://health.data.ny.gov/

Several sources cite FSC resource generated at approximately 1lb/inmate/day (FDEP, 2004; U.S. EPA, 1998). Data reported through the composting program of the NYDOCCS suggest that this estimate may be a high as their program averages about 0.65lbs/inmate/day (U.S. EPA, 1998). The estimate based upon NYDOCCS was used as it was assumed that these programs have high compliance rates and therefore the amount composted closely reflects the amount generated.

The State's five military bases were not included as this time. Although they house approximately 24,000 service persons there was insufficient information available to confidently derive a FSC resource factor at this time. Data on other governmental institutions was also excluded at this time.

Entertainment, hospitality and restaurants

Entertainment: consisted of amusement parks, golf courses, country clubs, ski and bowling facilities, museums, historic sites, parks, zoos, theatres, concert venues, racetracks and sporting arenas. These were identified through a marketing database query (Infogroup, 2014) based upon the following criteria:

- Primary NAICS 711219, 711212, 711310, 712, 713110,713910, 713920, 713950
- State of NY
- 10+ employees
- $1M+ in sales

FSC resource generation estimation was based upon the only reported study of this sector, which included audits of 53 California establishments (Calrecycle, 2014).

Hospitality: includes hotels, bed and breakfasts, Inns and other forms of lodging. A marketing database query (Infogroup, 2014) was made based upon the following criteria:

- Primary NAICS 721
- State of NY
- 10+ employees
- $1M+ in sales

FSC resources generated by this sector were based upon California audit data (Appendix (Calrecyle, 2006; Calrecycle, 2014) (Appendix A, Table A-8).

Restaurants were identified through a marketing database query (Infogroup, 2014) based upon the following criteria:

- Primary NAICS 722511
- State of NY
- 5+ employees
- $1M+ in sales

A FSC resource estimation factor based upon number of employees was used (Calrecycles, 2014).

Households

Most studies of household resources do not actually measure FSC resource generated but rather FSC resource disposed by auditing trash or MSW for a given population. Thus this estimate does not include resources that are backyard composted, disposed via in-sink garbage disposals or fed to household pets.

Estimated FSC resources generate were based upon the Calrecycle studies which averaged about 230kg/household/year (Calrecycle, 1999; Calrecycle, 2008). This was slightly lower than estimates gathered from a private community compost service that has collected data on weekly container pick-ups of approximately 200 households for 2 years (Appendix A, Table A-9). They report that most households were 2-person, but some were larger and some households had more than one collection per week. Therefore it is difficult to extrapolate this to statewide households. This is an area that would also benefit from further research. .

Table 2-4: FSC resource generation factors for FSC sector and sub-sectors

FSC sector and sub-sectors	FSC resource Generation Factor[a]	Units
Retail and Distribution		
Supermarkets	2000	kg/employee-yr
Convenience Stores	2000	kg/employee-yr
Big box stores	250	Kg/employee-yr
Food service and consumption		
Institutions		
Schools K-12	15	kg/student-yr
Universities	25	kg/student-yr
Community and grad schools	5	kg/student-yr
Hospitals	140	kg/bed-yr
Nursing Homes	140	kg/bed-yr
Facilities	100	kg/inmate-yr
Entertainment, hospitality and restaurants		
Entertainment	850	kg/ employee-yr
Hospitality	2100	kg/ employee-yr
Restaurants	1500	kg/employee/yr
Households	220	kg/household/yr

[a] Factors are rounded to the nearest significant digit

Summary of FSC resource generation factors

The foods supply chain resource generation factors used in this study for the retail and consumption stages are summarized in Table 2-4 below

2.2.2 FSC resource utilization pathways:

Several utilization pathways exist in the state. Data on resource utilization pathways was collected through surveys and reporting available publicly or accessed through a FOIL request as discussed below.

2.2.2.1 Donation

Resources can leave the FSC but still have the potential to be suitable for human consumption. This can include manufactured product that does not conform to specifications and excess supply that cannot be effectively marketed either due to appearance, damage to packaging or proximity to expiration data.

Ten regional food banks serve NYS.[15] A survey was sent to each of the food banks to gather data on the sources and composition of the FSC resources received (Appendix A). Additional data was gleaned from public sources when available. Only 3 of the 10 facilities were able to provide a detailed breakdown on the sources of FSC resources received and the data provided varied year over year. This data was averaged annually and scaled based upon the annual donations received to extrapolate it to the State level (Appendix A, Table A-11). The data is intended to serve as a starting point, with the suggestion that processes be put in place to improve future data collection.

While every effort is made by food banks to utilize the FSC collected some resources are not redistributed due to biological decay, health risk or capacity of the distribution channel and are diverted to other utilization pathways. Data on the amount

[15]Feeding America, Find a food bank, http://www.feedingamerica.org/find-your-local-foodbank/?_ga=1.164443327.161225226.1450706640

of non-distributed resources and the utilization pathways used was also solicited in the survey and was reported for 4 New York State food banks. On average 4% of resources received were estimated to be non-distributed. The composition and utilization of these resources varied across the food banks.

2.2.2.2 Diversion to feed animals and other beneficial uses

Feeding food scraps or food processing by-products to animals has been practiced for centuries. It is a way of returning resources back to the FSC providing nutrients and calories to animals and displacing alternative feeds. FSC resources can be directly fed (sometimes referred to as wet feed) to animals with minimal processing this is sometimes referred to as wet feed. FSC resources can also be processed on-site (usually including a drying process) or at another facility into a constituent that is sent to a feed mill and blended into commercial animal feed.

When FSC resources are used to substitute feed or another manufactured product they are put to beneficial use. In particular generators of FSC resources that are used in this way can be granted a beneficial use determination (BUD) from the NYSDEC. Once a BUD is granted these FSC resources are no longer considered wastes and are no longer under the jurisdiction of the Part 360 regulation of Solid Waste Management Facilities.

Additionally, the NYS Department of Agriculture and markets prohibits feeding "garbage" to cattle, swine or poultry. This prohibition is particularly aimed at avoiding feeding meat or animal parts to livestock. Thus garbage is often defined as "plate waste", prohibiting most food service and consumption phase resources to be fed to livestock. Meat scraps or trimmings from the food processing and retail sectors are also prohibited. FSC resources that can be fed to animals include dairy and cheese products or by-products, non-meat supermarket products, eggs, stale baked goods and discarded or scrap fruits and vegetables.

Data from BUD reports obtained from the NYSDEC provided information on FSC used to feed animals as well as other beneficial uses[16]. Although, generators seeking to divert food to animals are directed by the NYSDEC to seek a beneficial use determination, resources from smaller generators are often diverted to animals without any beneficial use reporting and thus are not included in this analysis. Also excluded are resources fed to animals on-site.

2.2.2.3 Anaerobic Digestion (AD)

Data was collected for three categories of Anaerobic digesters: 1) on-farm: manure based digesters, 2) POTW: publicly owned wastewater treatment plants (POTW) that employ anaerobic digestion, 3) and other: this included community digesters processing regional FSC resources as well as anaerobic digesters at food processing facilities when information was available.

On-farm: Several sources were used to identify on-farm AD facilities including maps available on the New York State Energy Research and Development Authority

[16] Provided by Gary Feinland, Environmental Program Specialist, Bureau of Waste Reduction and Recycling, NYS DEC via email April 27,2015

(NYSERDA)[17], U.S. EPAs AgSTAR[18] program and Cornell Dairy Environmental Systems websites.[19]

Facilities that import FSC resources require registration or permitting as solid waste management facilities (6 NYCRR Part 360). Data on FSC resources processed by these facilities was provided through the Organics Recycling Facilities Report which is maintained by the NYDEC, most of the data was for the calendar year 2012[20].

Solids or liquid effluent exiting the AD process was assumed to be returned to the agricultural phase through use as bedding, compost or as a land applied source of nutrients.

POTW: The American Biogas Council has compiled a list of POTW that utilize anaerobic digestion to treat wastewater.[21] Central data collection on facilities that import FSC resources could not be found. Information gathered through public sources and expert consultation was reported[22]. Treated effluent was assumed to be released to waterways and sludge landfilled or land applied.

Other: Information on AD facilities that import FSC resources that require registration or permitting as solid waste management facilities (6 NYCRR Part 360) was obtained through the NYS Organics Recycling Facilities (NYSDEC, 2015).

---

[17] New York State Energy Research and Development Authority, DG Integrated Data System, accessed 2015, http://chp.nyserda.org/facilities/index.cfm?Filter=ADG

[18] U.S. EPA, Livestock anaerobic digester database, accessed 2015, http://www.epa.gov/agstar/livestock-anaerobic-digester-database

[19] Cornell University College of Agricultural and Life Sciences, New York State Anaerobic Digester Locations, accessed 2015, http://www.manuremanagement.cornell.edu/

[20] 2014 annual reports for Beneficial Use Determination, Provided by Gary Feinland, Environmental Program Specialist, Bureau of Waste Reduction and Recycling, NYS DEC via email April 27,2015

[21] (ABC, 2014)

[22] Science Line, 2013; Biocycle, 2015; Leader Herald, 2015

Facilities that process FSC resources onsite such as food processors are not regulated in this way. Therefore it was difficult to estimate onsite wastewater treatment (WWT) from this sector, however in some cases data has been reported or is publicly available and is included in this study.

2.2.2.4 Composting

Many compost facilities are required to be registered or permitted as organics recycling facilities with the NYDEC (6NYCRR Part 360)[23]. Exempt from this regulation are household composting, crop residue or animal manure only composting and small composting facilities. Data on FSC resources processed by regulated facilities was obtained from the Organics Recycling Facilities Report and this supplemented with data provided in Planning Units Recycling Reports[24].

Sources that maintain maps of compost facilities in the state were also consulted as these often include relevant smaller facilities that may not be permitted (ie. those at schools).[25] In addition the NY State Department of Correction and Community Supervision (NY DOCCS) has an extensive compost program serving many Federal

---

[23] Title 6 Department of Conservation Part 360 Solid Waste Management Facilities, http://www.dec.ny.gov/regs/2491.html

[24] NYSDED, "2013 NYS Local Planning Unit Recycling Reports", provided by Gary Feinland, via email April 23,2015 and "2013 NYC Compliance Reports" provided by Chris Glander, Environmental Program Specialist, vie email April, 28, 2015

[25] Biocycle, Find a Composter, http://www.findacomposter.com/; Cornell Waste Management Institute, NYS compost Facilities Map (and surrounding states), http://compost.css.cornell.edu/maps.html;

prisons within the state. A list of these facilities and data on FSC resources processed was obtained from a report provided by the NY DOCCS.[26]

2.2.2.5 Land Application

Land application of organic material is a way to return valuable nutrients and help organically enrich soils. Facilities involved in land application of sewage sludge, non-sewage sludge, septage, food processing and other solid wastes may be subject to regulation under 6 NYCRR Subpart 360-4 Land Application Facilities[27]. A list of regulated land application facilities was obtained from the NYSDEC[28]. Certain FSC resources are not covered by this requirement and therefore were not included in this analysis. They include food processing wastes that are visually recognizable as part of a plant or vegetable, aquatic plant or fish hatchery waste or waste generated and treated on-site (such as pomace, stems or leaves) when applied below acceptable agronomic rates.

2.3. *Results*

2.3.1 Summary

Over 3.5 million tons of solid resources (average solids content approximately 30% solids) were estimated annually to be generated in the New York State FSC. An

[26] NYDOCCS, "NYDOCCS Compost Operations", provided via email from Tim Bender, Director of Resource Management, Correctioal Program services on April 24, 2014
[27] Land Application of Organic Waste, NYSDEC, www.dec.ny.gov/chemical/8797.html
[28] NYSDEC, "2011 Land Application of septage and non-recognizable food waste", provided by Christian Glander, Environmental Program Specialist, Bureau of Waste Reduction and Recycling via email, November, 19, 2013.

additional 22.4 million m^3 of low solids resources (usually under 3% solids content) were also estimated, from the food manufacturing and processing sector (Table 2-5).

Table 2-5: Summary of estimated post-harvest FSC resource generation

FSC stage	**Establishments**	**Solid resources (t/yr)[a]**	**Low solids resources[b] (m^3/yr)**
Primary production (post-harvest)	36,300	51,000	
Food manuf. and processors	1,092	777,000	22,426,000
Retail	4,366	353,000	
Consumption (out of home)	13,426	862,000	
Consumption (household)	7,234,743	1,592,000	
Total generation[c]		3,634,000	22,426,000

[a]Generally 30% solids or greater although some fruits or vegetables may have lower solid content, also packaged goods of any solids content.
[b] Generally 15% solids or less and often classified as “high strength” according to local POTW regulations, only assessed for manufacturers and food processors
[c] Total may not sum due to rounding

Several trends could be observed (Fig. 2-3) The earlier stages of the FSC tended to generate resources with more uniform characteristics resource heterogeneity increasing in the latter stages. The geographic distribution of the resources also tended to generally increase in latter stages of the FSC with number of establishments growing most dramatically to over 7 Million New York State households. The third trend observed was the decrease in utilization. Likely related to the increasing heterogeneity and geographic distribution, latter FSC resources showed lower rates of diversion and fewer FSC resource utilization pathways at this time.

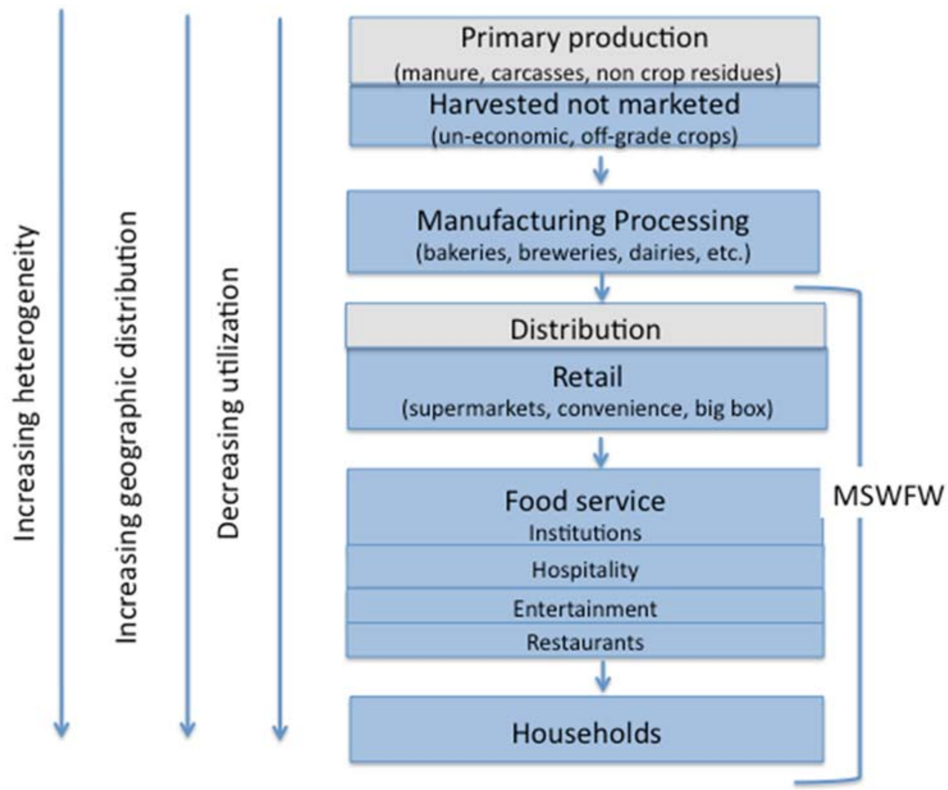


Figure 2-3: Trends observed (indicated by arrows) in resources generated in NYS FSC. Blue rectangles indicate segments included in this analysis. Grey rectangles indicate stages not included in this analysis.

Municipal solid waste food waste (MSWFW) includes solid waste generated from the Distribution and Retail, Food Service and Household sectors>.The consumption stage was responsible for the largest portion of solid FSC resources accounting for approximately 68% of solid resources, followed by food processors, retailers and primary production.

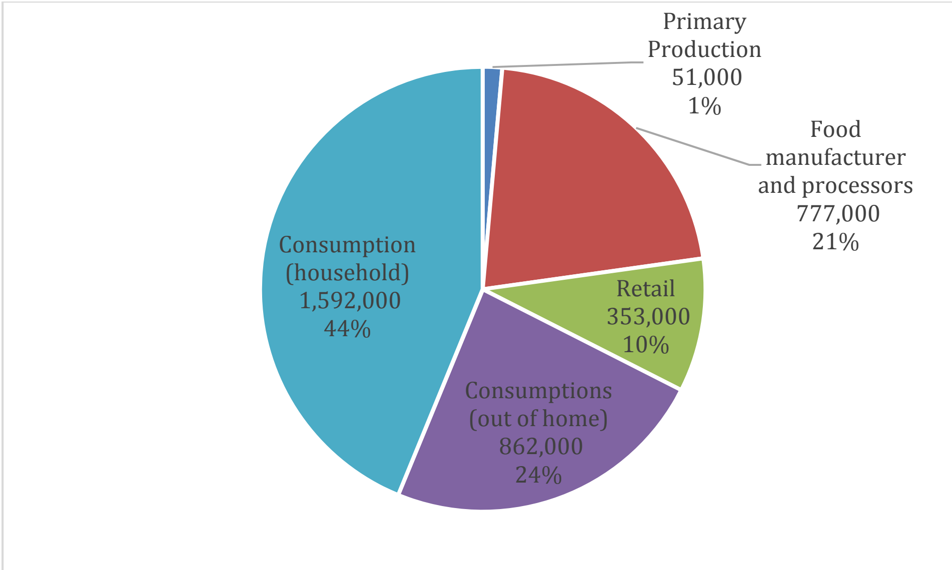


Figure 2-4: Post harvest resources (t/yr) and contribution by stage of FSC (%)

FSC resource generation is sometimes referred to as "hour glass" shaped because of higher generation at the beginning and ends of the food supply chain. This is particularly true when pre-harvest primary production resources are included. For example livestock manure from CAFOs was estimated to be 11,273,000 m^3 in NYS annually[29]. Although not included in this study these resources are also important to consider when planning a comprehensive FSC resource management strategy.

A summary of FSC utilization pathways is shown in Table 2-6.

[29] Organic Resource Locator, http://www.rit.edu/affiliate/nysp2i/organic-resource-locator-beta-version

Table 2-6: Summary of post harvest FSC resource utilization

Utilization pathways	**Facilities**	**Facilities currently utilizing FSC resources**	**Estimated solid resources processed (t/yr)**	**Estimated low solid resources processed (m^3)**
Donation	10	10	84,000[a]	-
Animal Feed / BUD[b]	669	16	84,000[b]	85,000[b,c]
AD[f]	181	20	-	434,000[d]
On farm	33	13	-	83,000[d]
POTW	144	3	-	97,000[d]
Other		3	-	254,000[d]
Compost[e]	222	68	55,000	-
Land Application	265	76	-	129,000
Total utilization			223,000	1,082,000

[a] FSC resources only, does not include food drives or walk-in donations
[b] Based upon 7 BUD reports available from the NYC DEC.
[c] Only includes FSC (ie. does not include corn ethanol production).
[d] Reported as volume (converted from gallons) although some solid wastes were utilized.
[e] Primarily retail and consumption out of home, does not include primary production, food processor on-site or household composting.
[f] Does not include some food processor on-site wastewaster treatment and land application

Approximately 2% of solid FSC resources were estimated to be donated. Programs targeted to connect FSC resource generators with local food banks are suggested to increase donation. In addition regional food banks should be connected with local utilization options. Regional food banks are uniquely positioned to coordinate utilization of FSC resources. In essence they can function as a MRF (materials recovery facility) for organic resources, gleaning what can be diverted to highest value (human consumption) and diverting the rest to industrial applications. One barrier to utilization of non-distributed food bank resources is that much of these resources are packaged. Therefore development into effective processes to handle these resources is suggested.

One of the barriers to food donation may be liability-related fear. The federal Bill Emerson Good Samaritan Food Donation Act (the "Bill Emerson Act" or "BEA") protects those who donate apparently wholesome food from liability except in cases of gross negligence or intentional misconduct. In fact a thorough review of reported conducted by students at the University of Arkansas, did not turn up a single case that involved food donation-related liability or any attempts to get around the protections offered by the BEA.

Of the 669 Beneficial Use Determinations identified 15 were for utilization of FSC resources. Data on the quantity processed was only available for 7 BUDs that process FSC resources. About half of those utilized whey from yogurt or cheese making as did two additional BUDs that did not have reported volumes. These resources were classified as low solids (ranging from 5% to 40%) and were used as animal feed (dry and wet), human dietary supplements and as fertilizer. Retail bakery waste processed as dry animal feed constituted about 40% of the reported beneficial use volumes. Brewery spent grains and retail waste fed directly to animals comprised the rest. BUDs granted without reported volumes also included using brine and alcohol distillate as de-icer, processed grapes to make tartaric acid and miscellaneous food processing resources as animal feeds and supplements or fertilizers. Resources utilized as animal feed or other beneficial uses are likely underestimated, due to the limited data on granted BUDs as well as the likelihood that some diversion of resources to animals is not reported.

Utilization of FSC resources to feed animals has the potential to avoid animal feed production and thus may be economically and environmentally preferable.

However, diversion of fresh resources to animals also presents challenges, as freshness, nutritional requirements and animal tolerance must be managed.

Approximately 55,000 t/yr of FSC resources were estimated to be composted. This was based upon 68 facilities, although a total of 226 compost sites were identified in the state when including those that process other materials (yard waste, carcasses, etc.). While assessing the State's capacity for composting is outside the scope of the present work, these facilities represent potential opportunities to increase FSC resource composting.

There were 181 anaerobic digestion facilities identified. Of the 144 POTW with an anaerobic digester, only 3 reported processing FSC resources; These facilities mostly co-digested resources from the food processing sector but some solid waste was also processed at facilities in central NY and NYC. Of the States 33 on-farm digesters, 13 reported co-digesting FSC resources, although volumes were only reported for 7 facilities. Two commercial digesters in the start up phase in Western New York, report that they will be dedicated mixed organics digesters, however all of the reported FSC resources in the "other"(not on-farm or POTW) category came from a digester located on-site at a single food processor. The number of food processors with on-site AD is unknown, however expansion of AD in the food manufacturering and processing sector should be explored. This should be guided by research to comprehend operational, environmental, social and economic impacts of co-digestion. .

Finally, 0.75 million m^3 of resources were estimated to be land applied. However, this is likely to be underestimated as this excludes on-site land application and application of recognizable food waste.

2.3.1.1 Primary production

Post-harvest FSC resources were estimated to be about 51,000 t/yr based upon the top NYS crops (Table 2-7). Apples and grapes were estimated to be the largest resources. Generally, grains and forages had zero loss factors whereas fruits and vegetables produced in NYS had loss factors ranging from 4% to 9%. The LAFA loss factors presented were qualified by USDA to be preliminary estimates and intended to serve as a starting point for additional research and discussion. Therefore, these factors were compared to data available for a few NY crops through the USDA National Agricultural Statistics Service (NASS) report of crops harvested and not sold (USDA, 2014). This comparison showed much lower quantities (0.6%-2% of crops harvested) than those calculated by the LAFA based factors. However, Matteson and Jenkins (2007) obtained an 8% factor for vegetable crops based upon a survey of California growers.

According to an NRDC report that interviewed large commercial vegetable and fruit growers and packers/shippers in Central California, culling for quality or appearance of harvested crops was the main reason for primarily production FSC losses (Gunders et al., 2012). One solution to this problem is to channel these products into cut or prepared products. The emergence of "baby cut carrots" is one example of market success with this strategy where carrots that don't meet consumer appearance standards are ground down to a smaller, more appealing product.

Table 2-7: Primary production level FSC resource estimation and comparison to reported "harvested and not sold"

		Comparison to 2014 harvested and not sold		Calculated using LAFA loss factor	
Crop	2011-2014 average annual harvest (t)[30]	Reported harvested not sold (t)[31]	Calculated % of harvest	LAFA loss factor (% of harvest)	Estimated FSC resources (t) [a]
Potatoes	4197.5			4%	9,000
Apples	1035.0	9.1	0.7%	4%	19,000
Grapes	183.3			9%	15,000
Pears	6.7			5%	300[a]
Peaches	5523.3	109.1	1.5%	5%	250[a]
Onions	2528.0			6%	8,000
Tart Cherries	7.9			8%	300[a]
Sweet Cherries	776.7	9.1	1.6%	8%	50[a]
Strawberries	3.6			8%	100[a]
Blueberries	3.5	0.02	0.6%	8%	100[a]
				Total	51,000

[a] Rounded to nearest 1000 except where doing so would result in zero reported.

Diversion of FSC resources back into the FSC is one form of source reduction. According to the EPA's food recovery hierarchy the next preferable utilization is donation to feed the hungry (Fig 2-5)[32]. The "Harvest for all" program reported that over

---

[30] Converted to metric tons and average of 2012, 2013 and 2014 data from:
National Agricultural Statistics Service (NASS), NY crop and livestock report, 2012,
http://www.nass.usda.gov/Statistics_by_State/New_York/Publications/Crop_and_Livestock_Report/2012/nycl1012.pdf
USDA NASS, 2014 State Agricultural Overview,
http://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=NEW%20YORK

[31] Harvested but not sold (Calculated for 2014 only): as reported in USDA NASS, "2014 State Agricultural Overview",
http://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=NEW%20YORK

[32] http://www.epa.gov/sustainable-management-food/food-recovery-hierarchy

4,362 t of primary production FSC resources (including meat and milk products[33]) were donated to NYS food banks in 2014 and estimate nearly 5,000 t donated in 2015.[34] This accounted for about 9% of estimated resources generated at this level. One enabler to greater donation from this sector is to provide a state tax credit for donation of locally grown food from farmers to food banks. Such a bill is currently in the NY State Senate.[35] Primary production FSC resources not diverted to humans are often diverted to feed animals or composted on-site. However, data on these pathways was not reported.

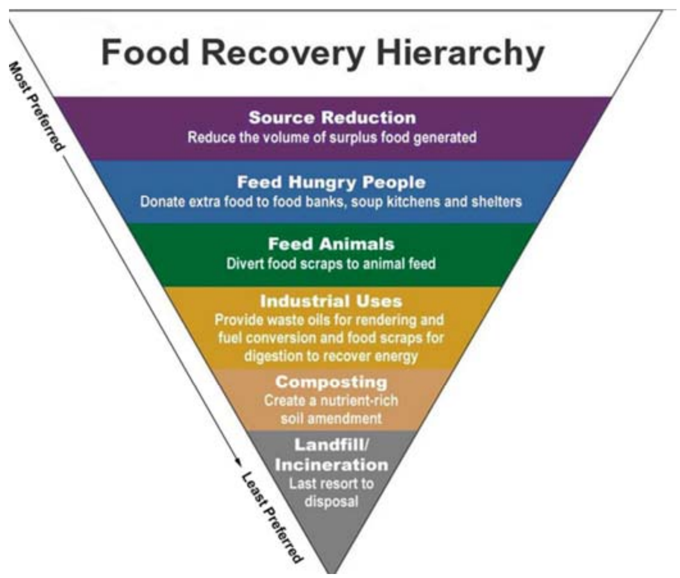


Figure 2-5: The EPA's Food Recovery Hierarchy prioritizes actions organizations can take to prevent and divert wasted food

---

[33] Although not included in this analysis, meat lost between primary production and retail is estimated to be 2%, seafood 0.5% and milk 0.25% of production respectively (NRDC, 2011).

[34] Farm Bureau of New York, "New York farmers donate record amount of food to regional food banks"; New York Farm Bureau kicks off 2014 State Annual Meeting with donation announcement; http://www.nyfb.org/img/topic_pdfs/file_6expuouf8b.pdf;
Farm Bureau of New York, "Every Farmer Investing in New York: 2015 State Priorities", http://www.nyfb.org/img/topic_pdfs/file_84xny0go5t.pdf

[35] NY State Assembly Bill A1812, https://www.nysenate.gov/legislation/bills/2015/a1812

2.3.1.2 Food manufacturers and processors

A total of 1,092 food manufacturers and processors were identified in the State, generating an estimated $32.7B in annual revenue (Fig. 2-6).

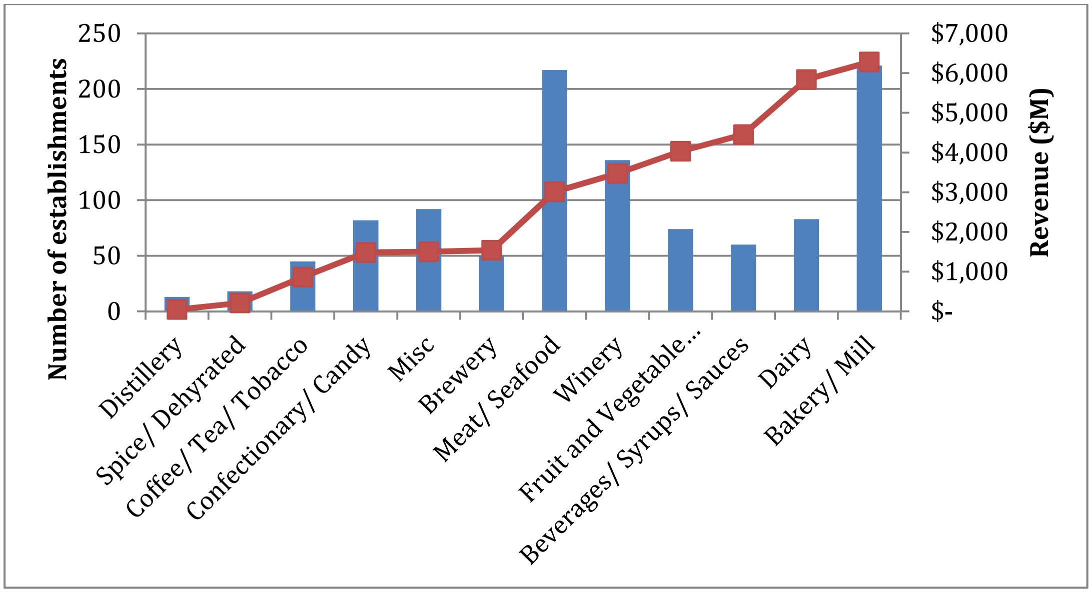


Figure 2-6: Manufacturer and food processor sector categories, number of establishments (bars) and revenue ($M) (red line).

Data was collected on 97 food manufacturers and processors through a combination of survey data and FOIL request and public reports.

Manufacturers and food processors reported using a variety of utilization pathways. FSC utilization was often complex, variable within a category and dynamic. The determination of which pathway to utilize fluctuated based upon economics and capability. For example, it was not uncommon for a large processors to send some resources off-site to AD or land application, treat some resources on-site, then to separate out solids and either divert it to animals, composting depending upon cost and availability.

The reported characteristics of the resources generated and utilization pathways for food processor solid resources is summarized in Appendix A Table A-9. A little less than half of the sample establishments (45 of 97) reported generating solid waste resources. Most was reported to be rejected product or scrap by-products. Reported utilization pathways included animal feed (wet and dry), composting, land application, beneficial use (phenolic recovery and rendering) and AD. A small amount was sent to the landfill which was reported to be packaged product. Confectionary/Candy, Dairy, Bakery/Mill and Fruit and vegetable processing generated the largest amounts of solid resources.

Low solids resources were reported to be generated by 61 of establishments but a significantly greater quantity of resources was reported (Appendix A, Table A-10). The resources consisted of wash water, liquid product and liquid by-products. Breweries generated the largest amount of low solid resources followed by dairy which showed lower average resources per establishment but had a large number of establishments in the sample. Caution was used in drawing conclusions on utilization pathways as the sample set may be biased toward POTW utilization, since high strength POTW discharges were a data source. However, although the sample constituted only 26% of the total revenue for this sector, it was a broad distribution and not dissimilar to the overall population (Appendix B Fig. A-11). Therefore in the absence of more data it was extrapolated to estimate 777,000 t/yr of high resources and 22,426,000 m^3 of low solid resources for New York State.

Resources generated and utilized on-site (i.e., composted, fed to animals, land applied) were not included in this analysis, which may understate the results. Most resources generated at this level do not go to a landfill. Low solid resources make up the majority of the resources generated at this level, indicating that a significant amount of water may be transported and treated which could potentially be reduced through investment into dewatering technologies. Larger generators tended to utilize many options for resource utilization, including on-site treatment and beneficial uses. Small manufacturers and processors tended to produce small quantities of waste and often lacked the resources and/or motivation to employ alternative utilization pathways. Therefore, efforts to share information and coordinate mid-sized processors may be beneficial. Utilization pathways were highly influenced by economics, which may increase vulnerability of utilization pathways as they compete for resources. Furthermore, utilization decisions based solely upon economics may not comprehend social and environmental impacts, which should be studied.

2.3.1.3 Retail and distribution

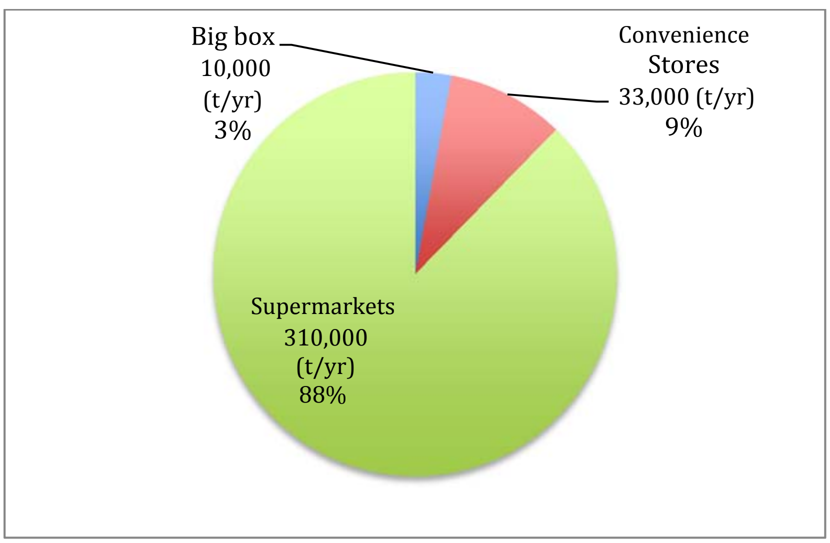


Figure 2-7 Distribution of FSC resources generated by the retail sector (t/yr) and contribution to total sector resource generation

The retail sector was estimated to generate over 350 thousand tons of FSC resources annually. Nearly 88% of this was estimated to come from Supermarkets (Fig. 2-7)

Large generators (estimated to generate greater than 100 t/yr) accounted for 73% of estimated FSC resources generated by this sector. There were 855 large supermarkets identified (Table 2-8).

Table 2-8: Number of establishments in the retail sector, estimated FSC resources generated from total stores and from large generators (>100t/yr)

Retail	**Stores**	**Resources (t/yr)**	**Stores > 100t/yr**	**Resources (t/yr) from stores >100t/yr**	**% Resources generated by stores>100t/yr**
Big box	167	10,000	19	2,000	24%
Convenience Stores	1752	33,000	5	1,000	3%
Supermarkets	2447	310,000	855	254,000	82%
Total	4,366	353,000	879	258,000	73%

Furthermore many supermarket chains operate several stores within the State. The top 20 supermarket chains in the state were estimated to represent 75% of both the estimated total resources (t/yr) and contained nearly 80% of the large generators (> 100t/yr) (Table 2-9)

Table 2-9: Top 20 supermarket chains in NY, number of stores, number of large stores and estimated resources per year for each chain.

Chain	**Number of stores**	**Stores > 100t/yr**	**Resources per chain (t/yr)**
Tops Friendly Market	171	139	38,000
Key Food	101	17	8,000
ALDI	99	0	2,000
Price Chopper	98	89	26,000
Associated Supermarket	73	2	3,000
C-Town	69	0	2,000
Save-A-Lot Food Stores	67	1	3,000
Shop Rite Supermarket	57	55	22,000
Hannaford Supermarket	53	48	15,000
Stop & Shop Supermarket	52	51	12,000
Super Stop & Shop	50	49	17,000
Waldbaum's	50	49	13,000
Wegman's	49	43	31,000
Pathmark	48	37	14,000
King Kullen	43	38	9,000
Gristede's Foods	38	2	2,000
A & P Food Store	37	32	8,000
Foodtown	32	6	3,000
Trader Joe's	20	18	3,000
Whole Foods Market	16	16	6,000
Total	1223	692	234,000
Percent of retail sector	28%	79%	75%

Total may not add due to rounding

FSC resources are generated at the retail and distribution level for a variety of reasons. Among the causes listed by Buzby et al., (2014) are damage to packaging, stale, spoiled or damage to products due to inadequate cooking or cooling, poor matching of supply to demand (including seasonal foods) and culling due to consumer preference. In France's "Inglorious" food campaign is one approach to reduce the amount of resources leaving the FSC for this reason. This program promotes off-grade produce as a new fad, appealing to consumer's sense of whimsy as well as their conscience and pocket books (at 30% less cost) (Grist, 2014).

Table 2-10: Estimated utilization of FSC resources (t/yr) from the retail sector.

Retail	Donated (t/yr)	Animal feed (t/yr)	Composted (t/yr)	AD (t/yr)	Landfill (t/yr)
Big box		7,996			2,000
Convenience Stores					33,000
Supermarkets		3,000	2,000	1,000	303,000
Total	145,00	11,000	2,000		325,000
% resources generated	9%	3%	1%	0%	87%

Data on FSC utilization at this level is scant, incomplete and uncoordinated. However, most resources generated at this level (about 87%) were estimated to be landfilled (Table 2-10).[36] About 9% was estimated to be donated based upon extrapolation of the data reported by 3 regional food banks operating in the state. This figure was lower than that estimated by the Food Waste Reduction Alliance (FWRA), an effort led by the Grocery Manufacturers Association (GMA), Food Marketing Institute, and National Restaurant Association, that collected data from 13 GMA members representing 30% of the U.S. revenue in the retail and distribution sector (BSR, 2013). That study reported an average of 17% of food waste was donated, but also large variation among respondents.

Diversion to animals was estimated to be about 3%. This was consistent with the FWRA study that estimated a 4% diversion to animal feed again noting large variation in responses. (BSR, 2013). A large portion of the diverted FSC resources were attributed to

[36] Resources not otherwise accounted for were assumed to be landfilled.

one big box store chain that diverted nearly 8,000 t of FSC resources to 14 NYS farms.[37] An additional 3,747 t from a supermarket chain was reported to be processed into a dry constituent for animal feed production[38].

Based upon the data gleaned from the NYS organics recycling reports and planning unit reports only about 1% was estimated to be composted annually from the retail sector. This consisted of resources from pockets (4 to 7 in a region) from four NYS supermarket chains.

One chain reported utilizing anaerobic digestion and this was estimated to be less than 1% of total retail FSC resources.

2.3.1.4 Consumption

About 65% of the FSC resources generated at the consumption stage come from households, 31% from entertainment, lodging and restaurants and only 4% from institutions (Fig. 2-8).

Most of the resources were generated from food service operations and consist of kitchen preparation waste, prepared but un-served foods or post-consumer plate waste. Food safety concerns and lack of logistics infrastructure make donation to humans or diversion to animals challenging for these resources although some options do exist. The most common alternative utilization for consumption phase resources was composting with AD of these resources emerging. However, most FSC resources generated at this level currently go to a landfill.

---

[37] NYS DEC, Annual BUD report 2014, provided by Department of Solid Waste Management.

[38] NYS DEC, Livestock Annual BUD report 2014, provided by Department of Solid Waste Management.

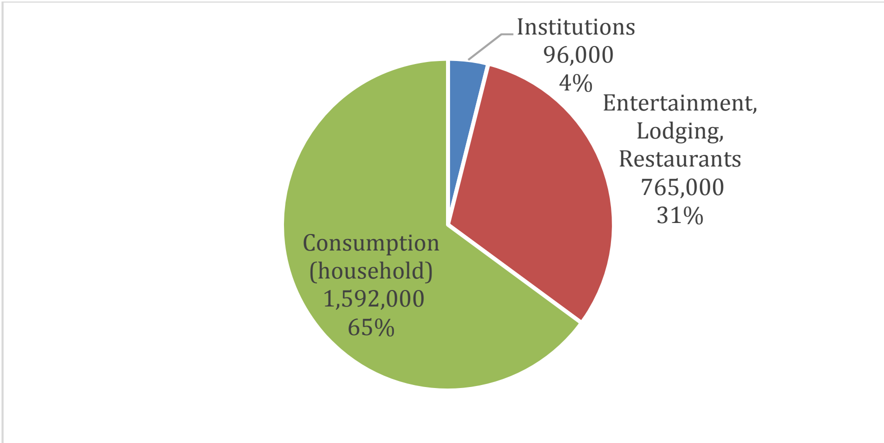


Figure 2-8: Estimated FSC resources (t/yr) and composition(%) of the consumption stage

Institutions

The education sector was estimated to generate 63%, of institutional FSC resources, with 26% estimated from the health and medical sector 10% from correctional facilities (Fig 2-9).

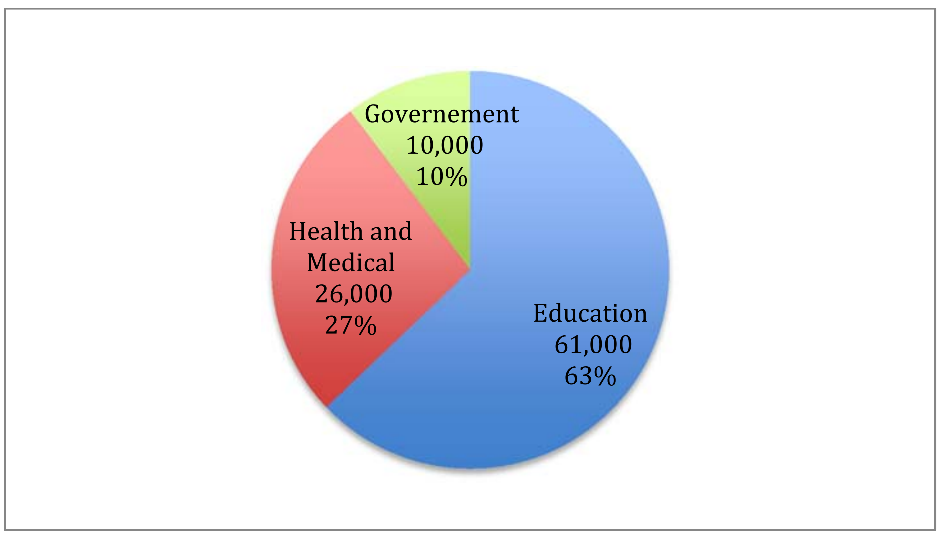


Figure 2-9: Institutional sectors, estimated FSC resources generated (t/yr) and share of total institutional sector resources generated (%).

About 50% of the resources come from 254 large generators; 197 large generators were in the educational sector (Table 2-11).

Table 2-11: Number of establishments in the Institutional sector, estimated FSC resources generated from total institutions, large generators (>100t/yr) and percent of resources generated by large generators (t/yr).

Institution	Institutions	Resources (t/yr)	Institutions > 100t/yr	Resources (t/yr) from institutions >100t/yr	% resources from stores>100t/yr
Education	1067	61,000	197	37,000	61%
Health and Medical	856	26,000	26	4,000	15%
Government	129	10,000	31	6,000	64%
Total	1,831	96,000	254	48,000	50%

Education: Large schools and school districts constitute more than half of the educational resources and generate about 37,000 t/yr (Table 2-12).

Table 2-12: Number of establishments in the education sub-sector, estimated FSC resources generated from total educational institutions, large generators (>100t/yr) and percent of resources generated by large generators (t/yr)

Education	Institutions	Resources (t/yr)	Institutions > 100t/yr	Resources (t/yr) from Institutions >100t/yr	% Resources from Institutions >100t/yr
K-12 schools districts	725	40,000	79	23,000	56%
Colleges and Universtities	342	20,000	118	15,000	73%
Total	1,067	61,000	197	37,000	61%

Despite the challenges of donation at this level of the FSC, the Food Recovery Network has diverted approximately 24 t/yr of resources from colleges and universities in NYS to feed the hungry[39](Table 2-13). Founded in 2012 the Food Recovery Network is a national student movement that currently operates at 10 NYS colleges and universities and is growing annually. Fourteen colleges report composting (4-year schools (11) and 2-year schools (3)) and 1 university was reported to send food waste from a dining facility to an anaerobic digester. Many educational institutions have environmental or social goals that support food utilization and waste reduction, despite the small impact these of these programs they can be viewed as seed locations to build awareness and infrastructure. Also since many of these facilities track their resource flows, a voluntary repository or reporting system could improve data availability on this sector.

Table 2-13: Estimated utilization of FSC resources (t/yr) from the Education sub-sector.

Education	Donated (t/yr)	Composted (t/yr)	AD (t/yr)	Landfill (t/yr)
K-12 schools districts		400		40,000
Colleges and universities	20	2,000	150	18,000
Total	20	2,400	150	58,000

Figures in table are rounded to nearest hundred except where value is under 100 when figures are rounded to nearest 10.

Health and medical: This sector was estimated to generate about 26,000 t/yr. Only 26 establishments were estimated to generate greater than 100 t/yr (Table 2-14).

---

[39] Data provided by Food Recovery Network, Fall 2014/Spring 2015, personal communication, May, 5, 2015. Note figures in table

Table 2-14: Number of establishments in the health and medical sub-sector, estimated FSC resources generated from total Health and Medical institutions, large generators (>100t/yr) and percent of resources generated by large generators (t/yr)

Health and Medical	Institutions	Resources (t/yr)	Institutions > 100t/yr	Resources (t/yr) from Institutions >100t/yr	% Resources generated by Institutions >100t/yr
Hospitals	226	10,000	23	4,000	35%
Nursing Homes	630	16,000	3	300	2%
Total	856	26,000	26	4,000	15%

Very little data was found to support alternative utilization pathways for this sector. While a few compost facilities reported accepting resources from this sector only one nursing home was identified by a facility that reported processed volumes (Table 2-15).

Table 2-15: Estimated utilization of FSC resources (t/yr) from the Health and Medical sub-sector

Health and Medical	Composted (t/yr)	Landfill (t/yr)
Hospitals		10,000
Nursing Homes	40	16,000
Total	40	26,000

Additional precision added to show data.

Government: There were 129 correctional facilities identified in NYS. Only 31 facilities were estimated to generate greater than 100 t/yr and account for nearly 80% of the estimated resources from this sector (Table 2-16). Most of the larger facilities are federal prisons.

Table 2-16: Number of establishments in the correctional sub-sector, estimated FSC resources generated from total correctional institutions, large generators (>100t/yr) and percent of resources generated by large generators (t/yr)

Correctional	**Institutions**	**Resources (t/yr)**	**Institutions > 100t/yr**	**Resources (t/yr) from Institutions >100t/yr**	**% Resources by Institutions >100t/yr**
County	64	2,000	2	-	36%
Federal	59	7,000	27	5,000	79%
NYC	6	1,000	2	1,000	99%
Total	129	10,000	31	6,000	65%

The NYS Department of Correction and Community Supervision (NYDOCCS) operates a very successful compost program. With operations at 24 of the State's federal prisons, it services a total of 47 facilities and composts nearly 80% of the food waste from federal prisons (Table 2-17).

Table 2-17: Estimated utilization of FSC resources (t/yr) from the Correctional sub-sector.

Correctional Facilities	**Composted (t/yr)**	**Landfill (t/yr)**
County		2,000
Federal	5,000	2,000
NYC		1,000
Total		10,000

Entertainment, hospitality and restaurants

This sector was estimated to generate nearly 760,000 t/yr of FSC resources. More than half of the resources were estimated to be generated by restaurants (Fig. 2-9).

However, additional data on FSC resource generation, particularly for the entertainment and hospitality sector would improve certainty.

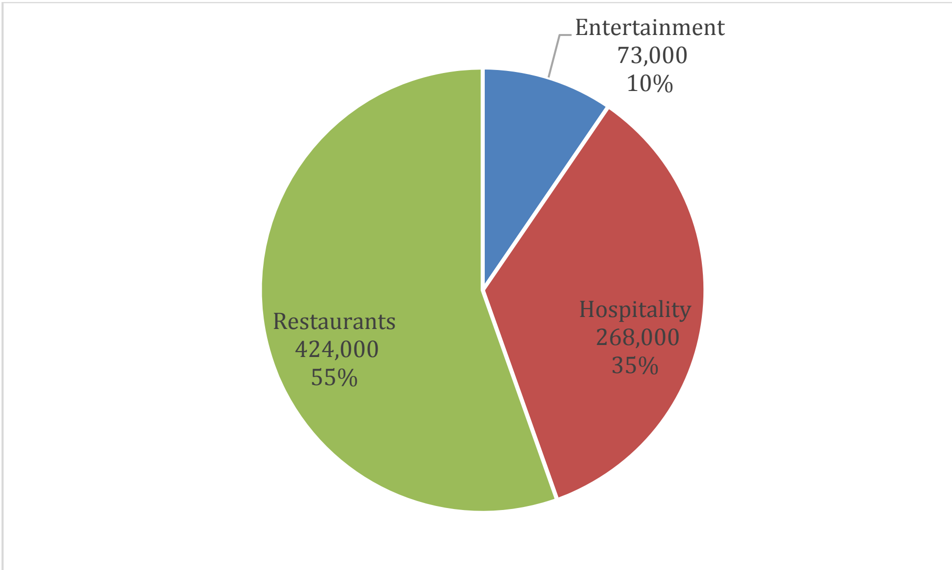


Figure 2-10: Entertainment, hospitality and restaurant sectors, estimated FSC resources generated (t/yr) and share of total sector resources generated (%).

Table 2-18: Number of establishments in the entertainment sub-sector, estimated FSC resources generated from total educational institutions, large generators (>100t/yr) and percent of resources generated by large generators (t/yr)

Establishments	**Establishments**	**Resources (t/yr)**	**Establishments > 100t/yr**	**Resources (t/yr) from establishments >100t/yr**	**% Resources generated by stores>100t/yr**
Entertainment	1,418	73,000	108	39,000	53%
Hospitality	2,017	268,000	580	141,000	53%
Restaurants	7,939	424,000	752	32,000	8%
Total	11,374	765,000	1,440	212,000	28%

Several large generators are in this sector along with many chains in the hospitality sector. However, nearly 8,000 restaurants were estimated to generate nearly 424,000 t/yr (Table 2-18).

Table 2-19: Estimated utilization of FSC entertainment, hospitality and restaurant (t/yr) from the Correctional sub-sector.

Sector	FSC resources (t/yr)	Composted (t/yr)	Landfill (t/yr)
Entertainment	73,000	2,000	71,000
Hospitality	268,000	2,000	266,000
Restaurants	424,000	80	424,000
Total	765,000	4,934	760,000

Additional precision added to show data

Although data reporting for this sector is limited, it is estimated that only about 0.5% of the resources generated at this level were composted (Table 2-19). Two historic Inns and two chain hotels were reported to compost FSC resources. There was little reported FSC resource utilization in the entertainment sector aside from the reported composting of food waste one large arena. A small NYS company is known to utilize resources from another venue to make animal treats, however this currently is only estimated at about 5 t/yr. Similarly, while some compost or AD facilities report processing restaurant resources there was very little data to quantify this utilization.

<u>Households</u>

Households were estimated to generate nearly 1.6 million tons of FSC resources. About 2% of FSC resources were estimated to be composted through a variety of mechanisms (Table 2- 20). Two entities reported MSW composting programs which utilized the largest amount of resources from this sector. Also reported were private collection programs, market drop off programs (such as GROW NYC) or facility drop-off programs. Also reported were collections either through private companies or non-profits at events such as races or festivals. Finally, resources with no explanation of source were allocated to this category.

Table 2-20: Household sector estimated utilization (t/yr)

Household sector	**Composted (t/yr)**
Events	0.2
Market collection	80
Area households/residents	200
MSW	24,000
Unknown	5,000
Total	30,000

Additional precision added to show data

2.3.2 MSWFW analysis

Resources leaving the retail and consumption stages are mostly solid waste and usually treated as municipal solid waste (MSW). The food scraps component of MSW is sometimes referred to as MSW food waste or MSWFW. Table 2-21 shows a comparison of the MSWFW resources estimated through the factor-based calculations resources to an estimate of MSWFW resources estimated bia reported MSW data and estimated MSW composition (NYSDEC, 2010).

Table 2-21: MSW FW generated in the retail and consumption stages. Number of establishments and estimated FSC resources generated.

Sector	Estimated FSCR (t/yr)	% of MSWFW
Retail	353,000	12%
Institutions	96,000	3%
Entertainment, Lodging, Restaurants	765,000	26%
Household	1,592,000	54%
Other	133,000	5%
MSWFW	2,939,069	100%

The difference amounted to about 5% of MSWFW, which was labeled as other and includes establishments not accounted for as well as estimation error.

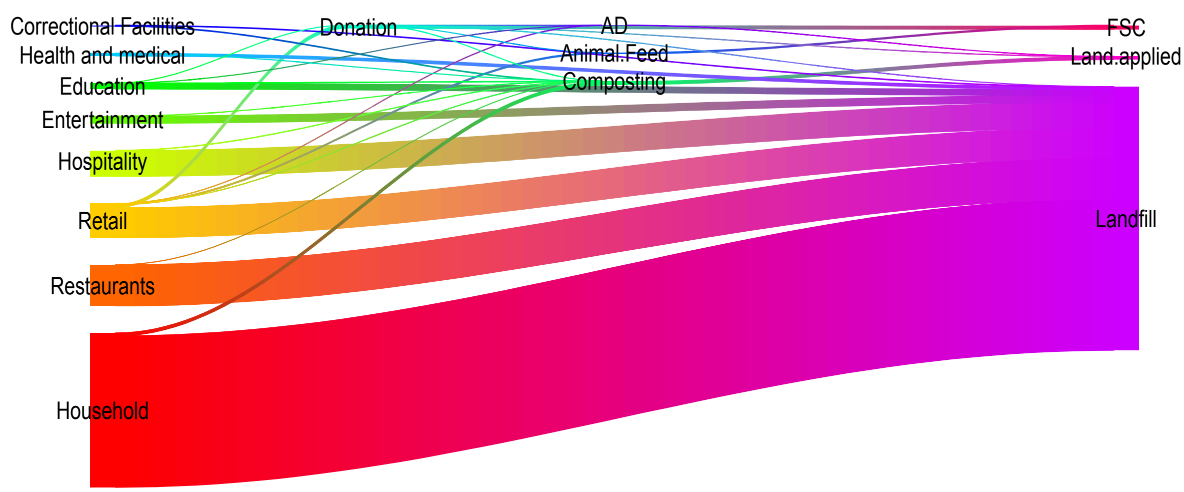


Figure 2-11: Sankey diagram of MSWFW FSC resources. The left side shows sources of FSC resourcs and the right side shows final treatment of those resources.

Resources generated by the Retail and Food Service sectors (excluding the residential and construction and demolition municipal solid waste) are sometimes referred to as Commercial FSC resources. This sector is often targeted for the early implementation stages of organic waste disposal regulation. Thus the NYS commercial sector generates approximately 41% of MSWFW (comprised of 29% from food service and 12% from retail)(Fig. 2-12). Restaurants account for more than 1/3 of the commercial FSC resources, retail establishments generate a little less than 1/3 and about 1/3 are estimated to originate in the hospitality (22%) and institutional (8%) sectors. The commercial sector contains 2,573 large generators (> 1t annually). These constitute 14% of commercial sector establishments and account for 43% of the resources from this sector.

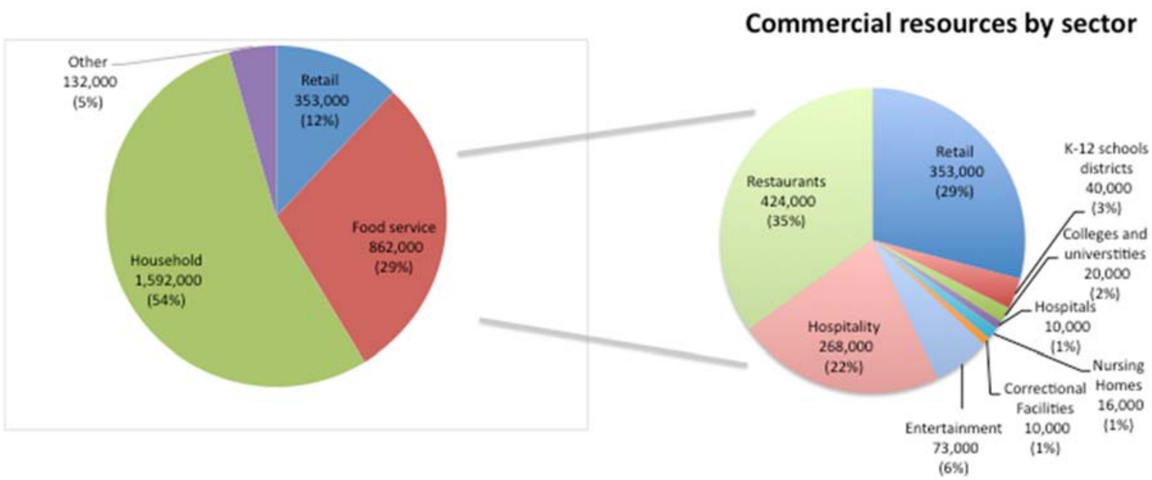


Figure 2-12: Commercial sector consisting of retail and food service sectors, broken down by type of generator (sub-sector) amount of FSC resources generated (t) and percent of commercial sector resources (t).

2.3.3 Geographical Information System (GIS) Analysis

In addition to the resource characterization material flow analysis presented geographic data has been collected. Geographic coordinates or addresses for FSC resource generators and utilization pathways along with linked data were loaded into a

GIS system (ArcGIS™) and manipulated into geospatial data sets with a unified format (ie. coordinate system, projection, datum, etc.) (Fig. 2-13).

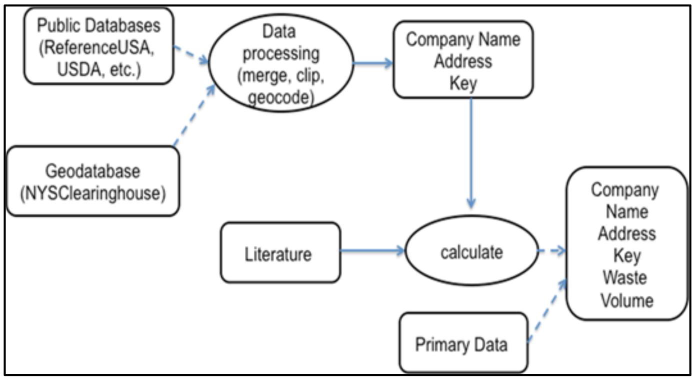


Figure 2-13: General Methodology used to develop geographical information system

The FSC resource generators and utilization pathways were organized into layers or map views (i.e., legend items) that could be toggled on or off as desired. This allowed for certain subsets of data to be displayed in a map view, (for example just dairy food processors and anaerobic digesters). Each entity was mapped as a point which when selected displayed a pop-up with a subset of information associated with that entity (i.e., name, type of resource/pathway, etc.)

Because management of FSC resources is highly dependent upon geography since resource generation is distributed and transporting FSC resources long distance is costly and problematic, this information has been made available as a web-based GIS tool, the Organic Resource Locator (ORL) (see Ebner et al., 2014b).

2.3.4 Limitations and future work

Data quality was poor, many different sources, manual entry of data, and conflicting reports make analysis challenging and results uncertain.

Several establishments were not included in the estimate. These included smaller establishments but also categories such as ice cream shops, farm markets retail bakeries, wholesalers/distributers and coffee shops, which may represent important sources of FSC resources. For example, based upon an estimated 3.1kg per capita consumption of coffee in the U.S., over 62,000 tons of coffee grounds are generated in NY State annually[40]. Also since many of these establishments market and distribute perishables such as baked goods or produce a quantifying the FSC resources from larger establishments or chains of stores within this sector is suggested as future work.

Another significant source of FSC resources excluded from this analysis are fats oils and greases. This includes used vegetable oil and grease trap waste, both of which are often collected at food service establishments. While there are several options to utilize these resources, how they are currently utilized in NYS is an area for future study.

A literature review has also shown that many of the resource generation factors were based upon dated and narrow studies. A detailed study of the NYS Educational sector showed that estimation factors based upon meal level audits were difficult to extrapolate to the institutional level. Therefore, establishment level waste audits were preferred for this purpose. Furthermore, distributed data from institutions performing

[40] Euromonitor International, "Coffee industry market reports 2015", http://www.euromonitor.com/coffee and US Census Bureau, 2015, "State and County Quick Facts", http://quickfacts.census.gov/qfd/states/36000.html

audits or monitoring diversion programs can be important sources of data. The breadth of this data may compensate for concerns over rigor vs. peer-reviewed studies. Other sectors with data gaps include the health and entertainment sectors.

2.3.5 Conclusions

This study provided a comprehensive analysis of FSC resources from post harvest through utilization. A set of resource generation factors and data has been provided to estimate FSC resources for New York State. The current state of resource utilization was also presented.

Food processors and manufacturers generated an estimated 22.4 m^3/yr of low solids resources and approximately 777 thousand t/yr of solid resources annually making this sector the largest source of resources in the post-harvest FSC. However, this sector utilized a variety of alternative pathways and very little was sent to landfill. The significant amount of low-solid resources generated suggest that efforts to reduce transporting or treating water should be explored through development of separation technologies. Opportunities to utilize large types of resources (i.e., dairy waste, brewery waste and fruit and vegetable processing waste) in beneficial uses and industrial utilization pathways should also be explored. While high and low solid resources were reported separately, it is worth noting that many facilities that operate traditionally wet or dry utilization processes are accepting other types of resources and mixing or treating them to achieve the desired process solids content. This suggests that the boundary between solid and liquid resources may be blurring and a data collection and implementation strategies should consider both high and low solid resources across the FSC. Finally, FSC utilization in this sector is dynamic and heavily influenced by economics. The lack of stability in this area should be taken into account by utilization technologies especially those that rely heavily on "tipping fee" revenue.

Development of de-pack technology and infrastructure has potential to reduce landfilling from the food processing and the retail sectors. Programs to target large supermarket chains can also have a significant impact in the retail sector.

The consumption stage was estimated to generate the largest quantity of solid FSC resources with most going to landfill. Household consumption was the largest source of resources estimated to comprise over 50% of MSWFW. Municipal compost programs reported the largest utilization in this sector although the impact was still small. The diversity of approaches to collect these resources for composting is also encouraging. Entertainment, hospitality and restaurants were estimated to generate about 26% of MSWFW with little diversion. This sector has received little attention and additional data is suggested along with a focus on large generators.

Although a relatively small contribution, Institutions have received a lot of attention and in many ways are well suited to be early adopters of organics diversion. The largest diversion was seen in the federal prison compost program. The educational sector also showed a lot of activity although still a relatively small impact.

In considering the food waste hierarchy, donation rates could be improved through education, legislation and coordination with utilization pathways. Diversion to dry feed processes has the potential to avoid many of the challenges associated with wet feed by handling a variety of resources and providing a stable, balanced. Research into economic and environmental impacts of this pathway is also suggested. Understanding the social, economic and environmental impact of anaerobic co-digestion is also important as this technology has large potential at the State's on-farm and POTW AD facilities.

Finally, since many options exist to utilize FSC resources data assessment of the environmental and social impacts of diverting specific resources from one utilization pathway can supplement economic considerations to enable sustainable choices. This is considered in more detail in Chapters 2 and 4 of this dissertation. A clearinghouse to facilitate data and communication among generators and utilization pathways is also seen as an enabler going forward. This is also an area of further development through the NYSP2I.

Chapter 3 Climate change impacts of food supply chain resource utilization technologies

Waste derived biofuels are one way to respond to growing pressures to divert FSC resources from landfill while simultaneously generating renewable energy. Utilizing FSC resources as sources of biofuel can also potentially improve the economics of these technologies through additional revenues in the form of “tipping fees”. For these reasons we have seen an emergence of waste derived biofuels in NYS. Early commercialization efforts provide an opportunity to study the important environmental impacts of these emerging technologies, including climate change impacts. Local implementation of these technologies can be strongly influenced by regional factors such as climate, regulatory environment/incentives and availability of feedstock making a local analysis particularly informative.

This chapter describes comprehensive lifecycle assessments for climate change impact of two emerging technologies to utilize FSC in New York State. Both are based upon primary data collected at New York State facilities.

The first which is covered in section 3.1, assesses an anaerobic co-digestion process based upon data from a facility located in Covington NY. As the largest on-farm digester in the State, the co-digestion facility studied is representative of the state-of-the art facility, co-digesting dairy manure and industrial food wastes which are common feedstock for the region. Recently at stable production and with extensive data availability through access to an online data collection system and a collaboration with

Cornell University, this facility was uniquely positioned to fill a knowledge gap critical to the State's future FSC resource utilization strategy.

The second (Section 3.2) is a waste-to-ethanol facility formerly in Rochester, NY that is no longer actively operating. The process is a second-generation biofuel, which utilizes food waste rather than agricultural feedstock. The NYS facility also pilot's an innovative small-scale, distributed production model. Its location in downtown Rochester, NY and open access to data allowed for a novel contribution to the literature and data to support "green development" in New York State.

Each section in this chapter follows a similar outline, beginning with an Introduction to provide background, motivation and objectives of the analysis, followed by a Methods section that details the lifecycle assessment methodology and inventory data sources. A Results section then presents quantitative analyses of lifecycle greenhouse gas emissions, compared to conventional treatment pathways for the FSC resources.

The results of both the anaerobic co-digestion and waste-to-ethanol studies have been published in peer-reviewed journals (Ebner et al. (2015b) and Ebner et al (2014a), respectively.

3.1 Lifecycle greenhouse gas analysis of an anaerobic co-digestion facility processing dairy manure and industrial food waste

3.1.1 Introduction

According to the U.S. Environmental Protection Agency (EPA), methane (CH_4) emissions from manure management contributed 53 T carbon dioxide equivalents (CO_2e) to total U.S. anthropogenic CH_4 emissions in 2012 (U.S. EPA, 2014). Moreover, between 1990 and 2010 they rose 68%, with dairy farm emissions increasing 115% during the same period (U.S. EPA, 2014a). The EPA attributes this increase, despite a general decrease in national dairy populations, to the shift toward larger dairy facilities which utilize liquid-based manure management systems (U.S. EPA, 2014a). Landfilling of solid waste and treatment of wastewater have also been large sources of anthropogenic CH_4 emissions, contributing 103 t CO_2e and 12.8 t CO_2e respectively to the national inventory in 2012 (U.S. EPA, 2014a). Anaerobic digestion (AD) has the potential to mitigate these impacts by effectively capturing and utilizing CH_4 emissions, and offsetting fossil fuel emissions.

Manure management via AD can also reduce odors and increase farm nutrient management flexibility, while AD of food waste can allow food waste generators to respond to increasing regulation of landfilling and land application of organics. Combining food waste with manure is particularly attractive as it often improves farm-based digester economics due to improved biogas yield as well as additional revenue in the form of "tipping fees" generated from importing food waste. For these reasons, AcoD has been promoted, particularly in areas with strong dairy and food processing industries, such as Upstate New York which currently has 33 on-farm digesters (of the approximately 244 in the US[41] (AgSTAR, 2014).

Many studies have been conducted concerning the environmental performance of biogas production with varied results and objectives (a review is contained in Appendix Table B-1). Some of the variation in results can be attributed to a lack of comprehensiveness where significant phases of the lifecycle were neglected. In two separate, comprehensive, comparative studies, Poeschl et al. (2012a, 2012b) and Börjesson and Berglund (2006, 2007) used data from literature to model a variety of state-of-the-art biogas production systems for Germany and Sweden, respectively. Their results showed that lifecycle impacts varied greatly and were significantly affected by the feedstock, reference system, size and operation of the AD facility and end-use technology. Dressler et al. (2012) compared three biogas plants in Germany and concluded further that regional parameters such as agricultural practices, soil and climate also influenced results. Thus, as Börjesson and Berglund suggested, environmental studies of biogas systems should be based on data referring to the specific local conditions valid for the actual biogas system (Börjesson et al., 2006)

This study analyzed climate change impacts for an anaerobic digester that co-digests manure and industrial food waste (IFW). Data on feedstock, digester operation and effluent properties were combined with regional parameters when available (e.g.,

---

[41] U.S. EPA, AgSTAR Database of Livestock Digesters, May 2015, http://www2.epa.gov/agstar/livestock-anaerobic-digester-database

climate and soil characteristics) to provide an estimate of GWP impacts for a state-of-the-art AcoD in the Northeastern U.S. Data collected through interviews was used to model a reference case, representing the business-as-usual food waste disposal and manure management practices *in lieu* of AcoD. This allowed for an analysis of the consequential impacts incurred. Results are reported on an annual basis and based upon the functional unit of one metric ton (t) of influent processed.

There are few peer-reviewed studies of the environmental impact of AcoD in the United States. Several case studies have presented calculations of impacts using GHG registry protocols, however portions of the lifecycle have been neglected, such as the feedstock reference case emissions, digestate storage emissions and fertilizer displacement impacts (Artrip et al., 2013; Bartram and Barbour, 2004; Bentley et al., 2010; Pronto and Gooch, 2010). Furthermore, they have often been modeled using theoretical assumptions such as number of cows rather than empirical data.

While comprehensive European studies exist, there are significant regional differences that affect environmental impact analysis. For example, common European feedstock of pig slurry and energy crops are not prevalent in New York State (NYS) where AD is primarily dairy manure based with a strong shift toward AcoD with IFW. Feedstock composition influences upstream impacts to transport and pretreat the feedstock as well as biogas production. Comparative studies have considered MSW and IFW feedstocks, but the reference cases have either been excluded (Møller et al., 2009) or modeled to reflect European disposal practices (i.e. incineration or composting; Börjesson and Burglund, 2006, 2007; Poeschl et al., 2012a, 2012b; Rodriguez-Verde et al., 2014) The disposal pathways for IFW feedstock in this study were reported to be land application, diversion to animals, wastewater treatment and wastewater treatment followed by landfilling. (Landfilling of organic waste is not banned in NYS at this time).

Thus one novel contribution of this study was inclusion of the impacts of diverting IFW for use in AcoD. In addition, a comprehensive analysis of a US on-farm anaerobic co-digester was conducted. In doing so, regional differences such as limited regulation of CH_4 releases, the use of open-air storage pits, regional electric grid mix and climate and soil conditions were considered. Emission factors and a detailed methodology were provided for use in analyzing similar implementations, plus gaps in national and regional factors were identified to guide future research.

3.1.2 Methods

A lifecycle assessment (LCA) methodology was applied considering both direct and indirect GHG emissions. Direct emissions consisted of CH_4 and nitrous oxide (N_2O) releases due to biochemical processes, as well as carbon dioxide (CO_2) emissions due to the combustion of fossil fuels. Biogenic CO_2 emissions, such as CO_2 released during biogas combustion, were considered part of the photosynthetic carbon cycle and not included (U.S. EPA, 2014a). Indirect emissions consisted of upstream emissions derived from the provision of energy or materials used in the process and products or services that were avoided as a result of the AcoD process, such as grid electricity or inorganic (commercial) fertilizer production. Emissions associated with the construction, maintenance and decommissioning of the AcoD plant were not included as these were previously reported to be <1% of gross (Meyer-Aurich et al., 2012). Global Warming Potential (GWP) impacts were evaluated in terms of CO_2e using the latest Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (AR5) 100-

year Global Warming Potential (GWP) factors of 28, 265 and 1 for CH_4, N_2O and fossil CO_2, respectively (IPCC, 2013).

The reference and AcoD scenarios are shown in Fig. 3-1 and described as follows:

• Reference Case: Liquid manure slurry was collected and stored in an uncovered earthen pit until land-applied (via surface spreading or injection) as organic fertilizer when weather, crop and field conditions allowed, following a comprehensive nutrient management plan. IFW treatment was modeled based upon the alternative treatment reported for each of the IFW feedstock. These included land application (84%), wastewater treatment plant (WWTP) followed by landfill (14%), landfill (1%), WWTP (1%) and diversion to feed animals (modeled as a sensitivity analysis).

• AcoD case: Food waste was transported to the AcoD facility, combined with manure produced on-site and fed into the digester. Biogas produced by the anaerobic digester was combusted to generate electricity, which was exported to the grid. The digestate was fed into long-term uncovered earthen storage and recycled to cropland as described above for manure.

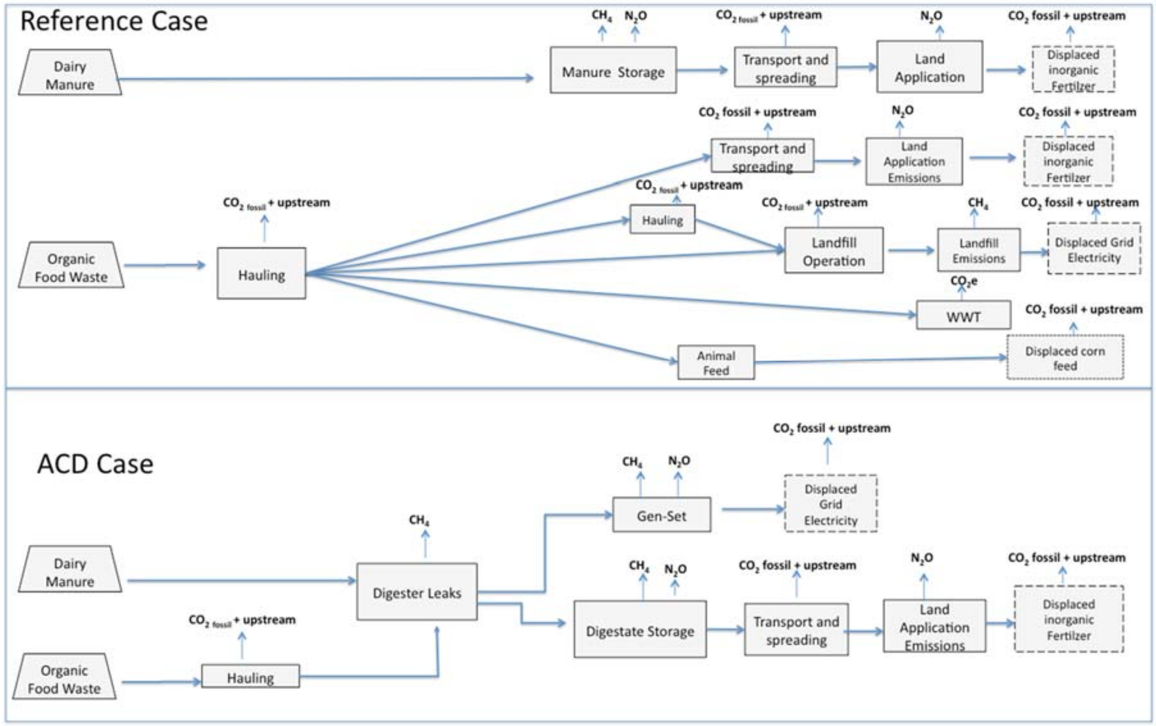


Figure 3-1: System boundaries and process flow for the reference and AcoD cases. Boxes represent individual process steps. Dashed boxes indicate a system expansion to include indirect emissions avoided due to displaced processes. Reference case emissions can also be considered an expansion to include avoided processes. Percentages shown in the reference case indicate mass composition of industrial food waste for each pathway. The symbol * indicates a pathway not included in the base scenario analyzed but used in the sensitivity analysis.

IFW, manure and digester effluent (digestate) characteristics as well as digester operational data (Table 3-1) were used to model the AcoD and reference cases. The data was based upon an on-farm AcoD in Western New York operating since January 2012. Data for the calendar year of 2013 was selected from a comprehensive monitoring study following the EPA ASERTTI reporting protocol and supplemented as needed, with additional detail found in the full report (Gooch and Labatut, 2014)). During the 12-month period under study, the AcoD facility blended 27% IFW with manure from approximately 1800 cows. The 8.3 ML, continuous stirred tank reactor operated at an average temperature of 41°C, with a hydraulic retention time (HRT) of 28 days and organic loading rate (OLR) of 2.1 kg VS/m3d. The process was multi-stage with a secondary biomass/gas storage tank with an approximate 4-day HRT. Electricity was generated using a 1.426 MW engine generator set. The system recovered 13% of the thermal energy produced from the biogas to provide heat to the process.

Because the objective of this study was to compare the impact of AcoD relative to alternative treatment of the same food waste and manure, unrelated factors were controlled to the extent possible. For example, although the farm under study switched from a flush manure handling system to a scrape system concurrent with AcoD implementation, both scenarios were modeled as scrape systems. Furthermore, while the farm utilized a screw-press separator, both reference and AcoD systems were modeled without solid-liquid separation, in order to utilize the data available and because solid-liquid separation can be implemented independently of AcoD.

Table 3 1: Key system parameters January 2013-December 2013[a]

System data	Representative value Jan 2013-Dec 2013 [a]	Units
Annual manure influent mass (t_M)[b]	88,247	t
Average VS content manure (VS_M)[c]	56.63	gVS/kg
Kjeldahl N manure (TKN_M) [c]	3,540	mg/kg
Bio-methane Potential manure ($B_{0,M}$)[e]	0.243	m^3 CH_4/t
Annual food waste influent mass (t_{FW})	32,024	t
Average VS content food waste (VS_{FW})[c]	193.50	gVS/kg
Kjeldahl N content food waste [c]	3,250	mg/kg
Annual total Influent Biomass (t_{IN})	120,271	t
Co-digestion ratio (v/v)	27:73	ratio
Annual digestate effluent mass (t_D) [d]	115,460	t
Average VS content digestate (VS_D)	30.37	gVS/kg
Kjeldahl N digestate (TKN_D)	3,097	mg/kg
Biogas methane content	58%	(%)
Methane utilized (Q_{CH4})	2,161,124	m^3
Annual electricity generated (MWh_{grid})	9062	MWh
Annual parasitic load ($MWh_{parasitic}$)	1101	MWh

[a] Based upon monthly data collection from the data set compiled by Gooch and Labatut (2014).
[b] Calculated from volume measurements using s.g=1.0 (Gooch and Labatut, 2014).
[c] Average of three readings including a supplemental reading collected for this study.
[d] Calculated based upon influent mass minus destroyed solids and water vapor (Gooch and Labatut, 2014).
[e] Based upon a total of 47 individual BMP assays performed on manure samples collected from six different NY dairy farms in various seasons reporting B_0=243±60 (L CH_4/kg VS)(Labatut et al., 2011).

GHG emissions were estimated throughout the process for both scenarios by combining the empirical data with emission factors gathered from literature as described in the following paragraphs (and Appendix Table B-2).

3.1.2.1 Reference case emissions

Dairy manure storage

Manure storage emissions were calculated per the IPCC methodology (Tier 3) for reporting of GHG emissions due to livestock (IPCC, 2006). CH_4 generated from the anaerobic decomposition of manure was based upon the volatile solids content (VS_M) and the bio-methane potential ($B_{o,M}$) of the manure, along with a methane conversion factor ($MCF_{i,,j}$) dependent upon the manure management system and climate. The $MCF_{ls,ny}$ for a liquid slurry management system in NYS was obtained from the U.S. GHG Inventory, which was modeled to include monthly temperature variation and account for monthly VS content of liquid slurry stored (U.S. EPA, 2014b).

Direct N_2O emissions result from the processes of nitrification and denitrification. These emissions were estimated as a portion of the total Kjeldahl nitrogen (TKN_M) stored using the IPCC default emission factor (EF_3=0.005) for dairy manure liquid slurry storage with a natural crust cover (IPCC, 2006; U.S. EPA, 2014b). Two sources of indirect emissions were calculated, indirect N_2O resulting from atmospheric deposition of volatilized nitrogen (primarily in the form of ammonia, NH_3) and indirect N_2O resulting from leaching and runoff. These emissions were calculated using the IPCC default emission factors (EF_4=0.01 and EF_5=0.0075, respectively) to estimate the portion of volatilized N ($Frac_{GASMS}$) or runoff/leached N ($Frac_{runoffleach}$) converted to N_2O-N (IPCC, 2006; U.S. EPA, 2014b). $Frac_{GASMS,ls}$ = 0.26 for dairy liquid/slurry management was taken from the U.S. Inventory of GHG emissions for NH_3 emissions (U.S. EPA, 2014b). The inventory used a $Frac_{runoff,ls,ma}$ = 0.007 for liquid/slurry management in the mid-Atlantic region derived from the EPA's Office of Water runoff data (as losses from leaching were stated to be small) (U.S. EPA, 2014b).

Land application of manure

Net GHG emissions from land application include the provision and combustion of fossil fuel to transport and spread manure, direct and indirect emissions due to subsequent biodegradation, emissions related to fertilizer displacement and long-term carbon sequestration. The emission factor reported by Møller et al. (2009) for transportation and field spreading based upon an average distance to the field of 20km was scaled to the 11km transportation distance reported in this study, resulting in an emission factor of 0.8 $kgCO_2e/t$ applied.

Direct N_2O emissions were determined by applying the default IPCC emission factor (EF_1=0.0125) to estimate the portion of N applied converted to N_2O, where N applied is the measured total N (TKN_M) minus N_2O losses (IPCC, 2013) and an additional 2% of N due to N_2 (Velthof et al., 2011) and NO (Stehfest and Bouwman, 2006) losses during storage and land application. Indirect N_2O emissions due to volatilization and leaching/runoff were also calculated for land application. Volatilization of N applied to land ($Frac_{GASM}$) is known to be affected by several variables including timing, application rate and application technique (Stehfest and Bouwman, 2006; Velthof et al., 2011). However, as US factors particular to these variables were not available, the IPCC default ($Frac_{GASM}$-0.20; EF_4=0.01) was used.

Precipitation and soil hydrological group data for Wyoming County, New York (U.S. EPA, 2014b) estimated a low probability of leaching therefore $Frac_{runoff}$ as described above was used to calculate indirect N_2O resulting from runoff/leaching.

Land applying dairy manure returns valuable nutrients to the soil thereby displacing inorganic (commercial) fertilizer use. Mineral-N is readily available for uptake by crops grown in the season of application; however it is subject to losses through NH_3 volatilization, denitrification and nitrate leaching. Organic N is more stable but over time is mineralized and becomes plant available. Inorganic fertilizer displacement was calculated using a mass balance approach to sum the N that will be available for plant uptake. Mineral-N, as measured via total ammonaical-N (TAN_M), was adjusted to subtract losses during storage and land application. This was added to 52% ($MinFactor_{,ny}$) of organic N that was estimated to be plant available within 3 years, based upon a mineralization profile for liquid dairy manure in NYS (Ketterings et al., 2003). Phosphorous (P) availability is assumed to be 90% of P applied (Risse et al., 2001) Potassium (K) displacement was not considered separately as it is often included in N and P inorganic fertilizer blends. GHG emission factors for fertilizer production were taken from the mean values reviewed by Wood and Cowie (2004). In addition to production emissions, displacing inorganic fertilizer displaces N_2O emissions associated with inorganic fertilizer application, replacing them with those of organic fertilizer. These emissions were calculated according to the IPCC protocol for indirect and direct N_2O emissions due to land application of inorganic fertilizer (IPCC, 2006). Finally, carbon in manure can be biochemically or biophysically stabilized in soil, resulting in carbon sequestration (CS). Risse et al. (2001) reviewed several studies of manure application and estimated 8-38% of C applied remained sequestered for temperate and frigid regions. A nominal value of 13% of VS is used in the present study. This value was chosen based upon the lignin content of manure (Labatut et al., 2011) as it was reasoned that while application rates, tillage practices, climate and crop rotation all affect carbon sequestration rates, over a very long time, the composition of the substrate has the largest influence on carbon remaining. Furthermore, this was consistent with the approach to estimating CS for landfilling, which was based upon substrate degradability experiments.

Food waste disposal

Log records maintained by digester personnel tracked the quantity and source of imported IFW. Most of the IFW (84%) was dairy processing waste, consisting of any combination of whey, wastewater, or milk products. Grease trap waste (GTW) and effluent from dissolved air floatation (DAF) wastewater treatment constituted 14%. The remaining 2% of the IFW influent was comprised of tomato processing waste and wastewaters from distilleries and wineries (Table 3-2).

Table 3-2: Food waste influent composition and alternative disposal pathways reported

Category	Percent of IFW influent	t/yr	Nominal scenario	All alternative disposal pathways reported
GTW	6%	1,999	WWT/landfill	WWTP/landfill, animal feed
DAF	8%	2,469	WWT/landfill	Land Application, animal feed, WWTP/landfill
Dairy processing wastewater and whey	84%	26,977	Land application	WWTP, Land Application, animal feed
Food processing waste (sludge)	1%	332	Landfill	Landfill, land application, animal feed
Other wastewater (distillery and winery)	1%	247	WWTP	Land app, WWTP
	100%	32,024		

Interviews with the waste generators or haulers were conducted to ascertain where the waste would have gone had it not been diverted to the AcoD facility. The predominate alternative disposal scenarios consisted of WWTP/Landfill disposal of GTW/DAF, land application of dairy processing waste and the remaining 2% split between landfill and WWTP (Table 3-2).

Land application of food waste

Land application of dairy processing wastewater has been practiced in the United States for over 50 years (Ghaly et al., 2007). The emissions associated with transporting the waste to a farm for land application were calculated using an average transport distance of 100km (USLCI, 2012). Emissions due to operation of farm equipment for spreading and direct and indirect N_2O emissions were calculated similarly to those described above in relation to manure. Farm spreading equipment emissions were calculated as described above in relation to manure. The portion of N that volatilizes (FracGASdairy) and that is leached (FracLEACHdairy) were estimated from studies of whey land application (Ghaly et al., 2007). Nominal estimates of N and P were derived from a survey of the literature (Table 3-3.) (Ghaly et al., 2007; Kushwaha et al., 2011; Watkins and Nash, 2010). Fertilizer displacement was calculated based upon 20% mineralization of organic N (Ghaly et al. 2007) and applying the emission factors for fertilizer production and fertilizer emissions discussed above. CS data was not available specifically for dairy processing waste, therefore it was estimated by applying data on the biodegradable fraction of dairy wastewater relative to that of dairy manure (Labatut et al., 2011) to arrive at 10% of VS applied.

Table 3-3: Reported dairy processing waste characteristics review

Description	VSdairy (g/kg)	TKNdairy (mg/l)	Pdairy (mg/L)	Source
Dairy industry, yogurt and buttermilk and cheese processing wastewater	N/A	14-830	9-280	27
Untreated cheese effluent, untreated whey	N/A	150-1400	42-640	28
Cheese whey	50	1820	468	25
Mixed diary processing waste effluent estimate used in this study[a]	50	800	400	

[a] Values used in this study are based upon the judgment of the authors and the descriptions in the literature above.
N/A not available

Table 3-4: Reported fats, oils and grease characteristics

Description	VS (g/kg)	Bo (ml/g VS)	Lo (m^3/t)	Source
GTW	107-252[a]	N/A		Baily et al., 2005
GTW	128-257	N/A		Razaviarani et al., 2008
GTW	170	845-928		Davidsson et al., 2008
GTW	158	900		Luste et al., 2010
GTW Estimation used[b]	182	887	161	
DAF	68	340		Luste et al., 2010
DAF	50	550		Woon et al., 2010
DAF Estimation used[b]	55.3	445	25	
Tomato seeds and skins	313[a]	218	68	Dinuccio, et al., 2010
Food Processing Estimation used[b]	313	298	20	

[a] Calculated from %TS and %VS/TS
[b] Values used in this study are based upon the judgment of the authors and the descriptions in the literature above. Food chain waste by nature is heterogeneous and varies based upon process, product and over time. A more detailed discussion of the effect of IFW characteristics can be found in Ebner et al., 2014

WWTP/landfill disposal of GTW/ DAF

Although waste haulers reported that GTW and DAF were disposed of at the WWTP, interviews with the WWTP operator revealed these wastes were actually combined untreated with wastewater sludge to achieve the solids content required for landfill disposal (Peletz, 2014). Thus the GHG emissions associated with the disposal of GTW/DAF included the impacts of transporting the waste to the WWTP, plus the treatment of the waste at the landfill. Transport emissions were calculated using a transport distance of 50km to the WWTP (USLCI, 2012). Landfill emissions were calculated as described in the following paragraph based upon the characteristics of GTW and DAF (Table 3-4).

Landfill disposal

Landfill emissions consist of those associated with fossil fuel used to collect the waste and operate the landfill, plus the net emissions due to the waste decay in the landfill. The emission factor for transport and operation of the landfill was taken from the EPA's WARM model (U.S. EPA, 2012). A multi-phased, first-order decay model was used to estimate CH_4 generation at the landfill. It was adapted from that used by the Climate Action Reserve (which is based on the UNFCC Clean Development Mechanism) to sum emissions over a 30-year rather than a 10-year period (CAR, 2011; UNFCCC/CCNUCC, 2008). A specific decay rate constant was not available for individual IFW constituents but one based upon experiments by de la Cruz and Barlaz (2010) for the broad category of food waste was used. Median bio-methane potentials from a survey of the literature (Table 3-3) were used for the various IFW (Baily, 2009; Davidson et al, 2008; Dinuccio et al., 2010; Luste and Luostarinen, 2010; Razaviarani et al., 2013; Woon and Othman, 2012). Landfill gas (LFG) captured was estimated using a gas capture factor (GC), representing the fraction of landfills in the State with LFG recovery systems (CAR, 2011; U.S. EPA, 2012) and a landfill capture efficiency schedule to model the efficiency of LFG collection over time (Levis and Barlaz, 2011). Electricity generated from recovered LFG was calculated based upon a conversion efficiency and plant capacity factor obtained from the EPAs landfill outreach program (U.S. EPA, 2012). Avoided grid emissions were calculated based upon the offset of non-baseload electricity generation, assuming the regional grid mix (U.S. EPA, 2010). Finally, 0.08kg C/kg dry food waste was estimated to remain sequestered in the landfill (Staley and Barlaz, 2009).

Municipal WWTP disposal of wastewater

Data on wastewater treatment emissions are limited and highly variable. The value of 0.518kg CO_2e/m^3 wastewater from the EcoInvent v.2.2 database was applied to the small percentage of wastewater that was diverted from a WWTP (EcoInvent Centre, 2007).

Diversion to feed animals

While the primary alternative treatment of dairy processing waste was reported to be land application, a sensitivity analysis was performed to analyze the impact of diverting dairy waste to feed cows. A transportation distance of 100km to the farm was assumed. Based upon the nutritional content of the dairy waste to the cows, 0.05 kg of corn was calculated to be displaced by 1kg of dairy processing waste (Chase, 2013). Displaced GHG impacts due to cultivation and production of corn/maize animal feed were obtained from the EcoInvent v.2.2 database (EcoInvent Centre, 2007).

3.1.2.2 AcoD case emissions

Food waste hauling

Delivery logs were used to calculate emissions associated with transportation of the food waste to the digester using the freight transport emission factor and the distance and the weight for each delivery (Table 3-5) (USLCI, 2012). A total of 1,537 trips from

15 waste generators were made, ranging from 22 to 194km, with a 40km average one-way transport distance and average payload of 22t.

Table 3-5: Summary of food waste delivery logs Jan 2012-Jan 2013

Waste Source	One-way distance (km)	# of Trips	total km traveled	Category	gallons
Source 1	34	227	7718	dairy	1,090,883
Source 2	55	156	8580	DAF	565,500
Source 3	62	71	4402	GTW	190,300
Source 4	64	18	1152	GTW	36,000
Source 5	72	109	7848	GTW	329,924
Source 6	78	16	1248	FPW	89,550
Source 7	194	26	5044	DAF	124,678
Source 8	17.91	366	6555.06	dairy	2,516,465
Source 9	56.1	2	112.2	other	3,410
Source 10	46.6	7	326.2	other	56,000
Source 11	22.43	127	2848.61	dairy	971,800
Source 12	37.24	401	14933.24	dairy	2,960,383
Source 13	36.56	8	292.48	other	12,950
Source 14	n/a	1		GTW	115
Source 15	n/a	2		GTW	2,200

Digester operation

Digester emissions consist of direct emissions due to leaks or incomplete combustion as well as indirect emissions offset by electricity generated. Canadian and German studies reported fugitive emissions ranging from 2.1% - 3.1% of CH_4 utilized (Flesch et al., 2009; Liebetrau et al., 2013). The nominal value of 3% of gas utilized was used. However, Liebetrau et al. (2013) noted that when leaks and malfunctions were eliminated, near zero fugitive emissions were measured. Conversely, automatic releases of biogas through emergency vents due to over-pressure conditions in the reactor or when flaring was not possible were observed. Therefore, a sensitivity analysis was performed using the IPCC default uncertainty range of 0-10% (IPCC, 2006). This range also allows for consideration of emissions due to flaring of biogas which were minimal during the period of study due to issues related to flare operation, but were reported to be on average 21% of gas produced in a study of seven NYS AD plants (Gooch et al., 2011). Site supplied measurements of gen-set exhaust reported 1,314 ppmv dry CH_4, which equated to 2.5% of the CH_4 utilized. This was consistent with reported values for incomplete combustion, which ranged from 0.4%-3.28% (Flesch et al., 2009; Liebetrau et al., 2013). N_2O exhaust emissions were a smaller contribution at 0.03g N_2O/m^3CH_4 utilized, which is also consistent with the range reported in the literature (0.02-1.75g N_2O/m^3 CH_4 utilized)(Flesch et al. 2011; Liebetrau et al., 2013)

Excess electricity beyond a parasitic load to operate pumps and mixers of about 12% of electricity generated was exported to displace grid electricity. Avoided emissions were calculated based upon a non-baseload emissions factor from the U.S. EPA eGRID database for the Northeast regional (NPCC) grid mix (U.S. EPA, 2010).

Digestate storage

Similar to the storage of manure, uncovered storage of liquid digestate can generate CH_4 over time. It has been shown that CH_4 emissions due to storage of digested manure are lower than those of raw manure due to VS destruction during the digestion process (Clemens et al., 2006; Nielsen et al., 2008). However, it has also been shown that just as co-digestion of substrates with manure increases biogas production, co-digested slurries show higher residual CH_4 emissions than manure-only slurries (Clemens et al., 2006). The main factors influencing digestate residual emissions are VS content, degree of degradation and storage temperature (Hansen et al., 2006; Menardo et al., 2011). While data specific to US conditions was not available, several European studies of digestate emissions were reviewed. Hansen et al. (2006) observed that temperatures in storage tanks directly fed from digesters were mainly affected by effluent temperature and ranged from 20°C-40°C. Batch studies of European AcoD samples incubated in this range had a mean value of 0.054 m3CH_4/kgVS stored (Table 3-6). This equates to 1.6 m3CH_4/t digestate which is consistent with the results of a study of 61 AcoD plants in Germany which reported average residual CH_4 potential of 1.5 m3/t digestate for multi-stage processes (Lehtomäki et al., 2008).

Table 3-6: Published studies of digestate storage methane emissions

Description / Source	kg VS/kW digestate	m^3 CH^4/ kg VS	Temp. (°C)	OLR (kgVS/ m^3-d)	HRT (days)	Feedstock
Non separated digestate (Gioelli et al., 2011)[a]	7.1%	0.034	41	1.4	105	Cattle slurry (12%);FYM (31%); Poultry manure (8);Maize silage (27%);maize residue (21%);Rice chaffs (1%)
Separated digestate (Gioelli et al.. 2011)[a]	3.3%	0.040	41	1.1	130	Cattle slurry (33%) ;FYM(24%); Maize silage (26%);Triticale silage (11%);Drying maize residue (3%); Kiwi (3%)
Sample A (Menardo et al., 2011)[b]	7.4%	0.038	41	2.25	105	manure(70%):energy crops (30%):IFW (10%)
Sample C (Menardo et al., 2011) [b]	2.5%	0.004	41	0.96	100	manure (37%);energy crops (47%);IFW (16%)
R2 @ 20C (Lehtomaki et al., 2008)	2.2%	0.076	35	2	20	manure(70%); sugar beets(30%)
R3 @ 20C (Lehtomaki et al., 2008)	2.3%	0.073	35	2	20	manure (70%); grass (30%)
R4 @ 20C (Lehtomaki et al., 2008)	2.7%	0.073	35	2	20	manure (70%; straw(30%)
Hansen et al., 2006	0.8%	0.068	55	N/A	15	MSW
This study estimate	3.0%	0.054	41	2.1	28	manure(70%); IFW(30%)

[a]Calculated from reported biogas produced, biogas concentration and VS content.
N/A not available

Modeling of digestate nitrous emissions is complex and influenced by many factors. Although reduction in organic matter typically prevents formation of a surface crust, which is associated with lower N_2O formation, several studies have reported increases in digestate N_2O storage emissions relative to untreated manure (Clemens et al., 2006, Amon et al., 2006). Therefore, digestate direct N_2O emissions were calculated using the IPCC default emission factor (EF_3) for manure storage.

It has been argued that emission factors based upon mineral-N rather than total N more closely model the volatilization and leaching/runoff processes (Velthof et al., 2011). Furthermore, the digestion process increases mineral-N content. However, in this case although mineral content of the feedstock was increased during digestion, the mineral content of the digestate (TAND) was similar to that of raw manure (TANM) (Appendix Appendix B, Table B-3). Thus, while emissions modeling based upon mineral-N content may provide a more accurate estimation, due to using the IPCC (TKN based) methodology will be comparable for both the AcoD and reference cases and thus have minimal impacts on net results. It has also been suggested that elevated pH and lower dry matter content, as found in digestate, may be conducive to higher volatilization. However, it is difficult to distinguish the magnitude of these effects vs. the impact of higher TAN content in studies of digested manure vs. undigested manure and studies of AcoD are lacking. Therefore, the IPCC default factors were also used to calculate indirect N_2O emissions and uncertainty analyzed (IPCC, 2006; U.S. EPA, 2014b).

Land application of digestate

Importing food waste increased the volume of organic fertilizer being land applied (Appendix B, Table B-3). This resulted in increased transportation distance to the fields for spreading from 11km for raw manure to 19km for AcoD digestate. The emission factor provided by Møller et al. (2009) was scaled and applied to calculate transport and spreading emissions.

Despite observing elevated pH and lower organic matter content, field experiments by Amon et al. (2006) and Clemens et al. (2006) observed no significant difference in N_2O as a percentage of mineral-N during land application of digested manure vs. untreated manure. Therefore, direct and indirect N_2O emissions, N losses and fertilizer displacement were modeled using the IPCC methodology as described for manure and analyzed through sensitivity analysis.

Little data exists concerning CS for AcoD digestate. Bruun et al. (2006) used an agronomic model to analyze inorganic fertilizer supplemented with digestate from MSW vs. composted MSW and observed that as time increased, the difference between CS rates between the two treatments decreased resulting in nearly identical rates after 100 years. Therefore 12% of carbon applied was used to model digestate CS which was the weighted average of the raw manure CS rate and the IFW CS rate.

3.1.3 Results and discussion

3.1.3.1 Comparison of reference case to AcoD case

Annual climate change impacts and emission factors per ton processed for the reference case and the AcoD case were compared (Table 3-7). It is important to consider that the impacts of a given food disposal and manure management pathway can be displaced by those of an alternative pathway, but the treatment of manure and food waste must be achieved. Thus the reference case can be considered a system expansion to account for the processes displaced by AcoD.

Annual net climate change impacts were reduced by 4,512 t CO_2/yr or 37.5 kgCO_2e/t influent treated. This is a 71% reduction for the AcoD case relative to the reference case. Displacement of grid electricity emissions was the largest contribution (avoiding 4,347 t CO_2e/yr or 35.3 kg CO_2e/t influent). The benefit of avoiding alternative IFW disposal (1,926 t CO_2e/yr or 16.0 kg CO_2e/t influent) was much greater than the impact of hauling food waste to the digester (129 t CO_2e/yr or 1.1 kg CO_2e/t influent). This was driven by GTW/DAF which avoided WWTP/landfill emissions (747.0 kg CO_2e/t GTW/DAF), although these only constituted 4% of the total influent. Impacts of digestate storage relative to manure storage resulted in (14.7) kgCO_2e/t influent, where the net benefit of lower VS overcame the increase in digestate volume due to imported IFW. In both cases, land application resulted in a net benefit with fertilizer displacement and carbon sequestration benefits offsetting direct and indirect fossil fuel and N_2O emissions. Land application of digestate had lower net benefit than that of manure due to greater volume being land applied, increased transportation distance to the field and lower carbon sequestration in the AcoD process.

Table 3-7: Summary of climate change impacts for a unit of waste (kg CO2e/t waste), annually (t CO2e/yr) and normalized by mass of influent processed (kg CO2e/t influent) for the reference and AcoD cases.

Reference Case				AcoD Case			
Process phase		Climate change impact per functional unit		Process phase		Climate change impact per functional unit	
	$kgCO_2e$/ t waste[a]	tCO_2e/ yr[b]	$kgCO_2e$/ t influent[b]		$kgCO_2e$/ t waste[a]	tCO_2e/ yr[b]	$kgCO_2e$/ t influent[b]
IFW disposal	60.1	1,926	16.0	IFW transport	4.0	129	1.1
Dairy waste	0.2	7	0.1				
GTW	747.0	1,493	12.4				
DAF	155.5	384	3.2				
Food processing waste	127.1	42	0.4				
Wastewater	0.5	0	0.0				
				Digester/gen-set emissions		2,243	18.6
				Displaced grid emissions		(4,247)	(35.3)
Manure storage	73.2	6,463	53.7	Digestate storage		4,691	39.0
Manure land application	14.6	1,286	10.7	Digestate land application		1,543	12.8
Displaced inorganic fertilizer	(10.7)	(946)	(7.9)	Displaced inorganic fertilizer		(981)	(8.2)
Carbon sequestration	(27.0)	(2,382)	(19.8)	Digestate carbon sequestration		(1,543)	(12.8)
Net reference emissions		6,348	52.8	Net AcoD emissions		1,836	15.3
					Net impact[c]	(4,512)	(37.5)
					Reduction	71%	

Positive values indicate emissions, negative values ( )indicate a reduction in emissions.
[a] Emissions associated with a the treatment of a single waste stream/manure.
[b] Emissions based upon the combined co-digestion influent processed.
[c] Net impact considers the replacement of the Reference process by the AcoD process.

The largest source of direct emissions was CH_4 (5,778 t CO_2e/yr or 48 kg CO_2e/t influent for the AcoD case and 7,602 t CO_2e/yr or 63.2 kg CO_2e/t influent for the reference case). N_2O direct and indirect emissions contributed 2,535 t CO_2e/yr (or 21.1 kg CO_2e/t influent) in the AcoD case and 2,210 t CO_2e/yr (18.4 kg CO_2e/t influent) for the reference case. N_2O emissions were larger in the land application phases than during storage and direct emissions were larger than indirect (Fig. 3-2). Direct fossil fuel emissions had a minor impact. Carbon sequestration offset emissions 1,543 t CO_2e/yr or (12.8 kg CO_2e/t influent) for the AcoD case and 2,879 t CO_2e/yr (or 23.9 kg CO_2e/t influent) for the reference case. Avoided fossil fuel use also contributed an offset (5,228 t CO_2e/yr or 43.3 kg CO_2e/t influent) for the reference case and 1,284 t CO_2e/yr (or 10.7 kg CO_2e/t influent) for the reference case.

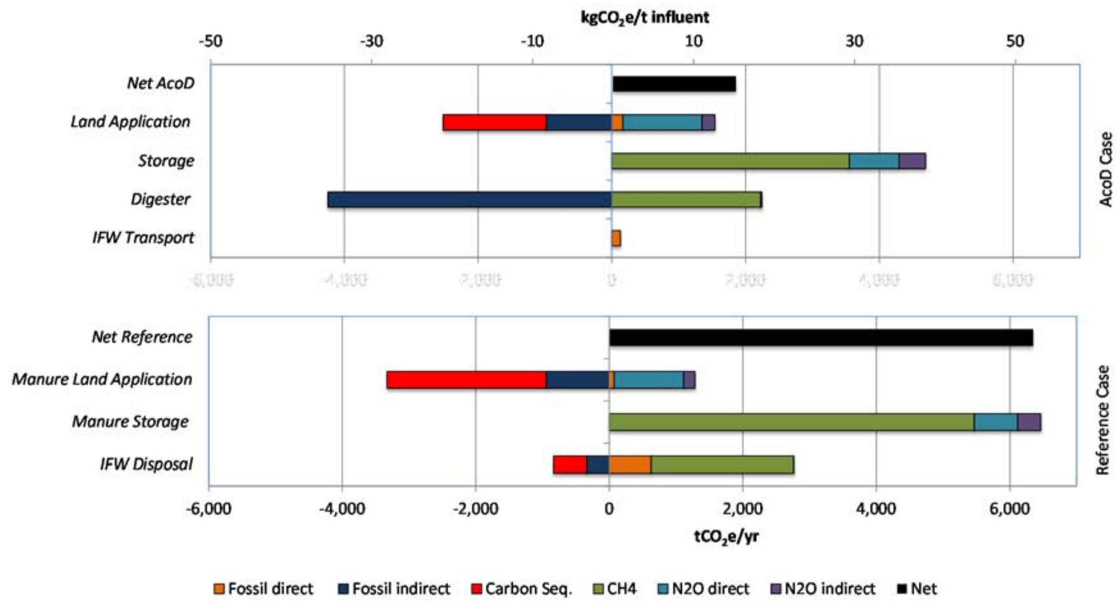


Figure 3-2: Contribution of greenhouse gases to climate change impacts annually (t CO_2e/yr) and based upon mass of influent processed (kg CO_2e/t influent) for phases of the reference and AcoD cases. Top black bars represent net emissions for each case.

3.1.3.2 Impact of feedstock composition

A sensitivity analysis modeled three scenarios where IFW composition and alternative disposal treatment varied. Reference case emissions and biogas production were estimated based upon the characteristics of the feedstock and biogas utilization and electricity conversion efficiencies calculated in this study were applied (as explained in Ebner et al., 2015a). Net AcoD benefit varied significantly (Fig. 3-3a-d). Highly degradable GTW co-digestion increased the net benefit by an order of magnitude (Fig 3-3b) to (29,969) t CO_2e/yr or (249.2) kg CO_2e/t influent. This was due to avoidance of significant landfill emissions as well as an increase in displaced grid electricity with only a minor increase in fugitive emissions. Diverting whey from feeding animals (Fig. 3-3d) resulted in the lowest net benefit (1,030) t CO_2e/yr or (8.6) kg CO_2e/t influent. While benefits of avoiding raw manure storage still enabled a net benefit, diverting whey from feeding animals incurred emissions due to production impacts of replacement feed, with whey feedstock providing only moderate CH_4 production.

Table 3-8 shows the estimated impact of avoided landfill emissions for the individual IFWs per ton.

Table 3-8: Landfill emissions per t source feedstock

	Landfill emissions (kgCO_2e/t source)		
	GTW	DAF	FPW
Landfill Operations ($EF_{LF,OP}$)	44.00	44.00	44.00
Landfill methane emissions ($EMLF_{CH4}$)	825.82	128.23	102.59
Grid displaced emissions	(114.97)	(17.85)	(14.28)
Carbon Storage	(13.19)	(4.22)	(5.19)
Net landfill emissions	741.66	150.16	127.12

3.1.3.3 Impact of CH_4 losses

Uncertainty in estimating CH_4 storage emissions can have a large impact on the results (Fig. 3-3h, Fig. B-1, Fig. B-2). However, nominal factors for CH_4 emissions resulted in a loss of 8.8% of CH_4 utilized. Capturing CH_4 generated during storage eliminates atmospheric emissions and displaces grid emissions. This would more than double net AcoD benefit to (9,526) t CO_2e/yr or (79.2) kg CO_2e/t influent (Fig. 3-3e).

Similarly, nominal fugitive emissions were modeled as 3% of CH_4 utilized. Two scenarios explored the impact of uncontrolled CH_4 releases and leaks. If CH_4 leaks were reduced to zero, the net emissions would be reduced to (47.6) kg CO_2e/t influent (Fig. 3-3f). However, emissions of 10% are quite possible through poor system inspection and uncontrolled releases, which would reduce the net emissions to (13.9) kg CO_2e/t influent (Fig. 3-3g).

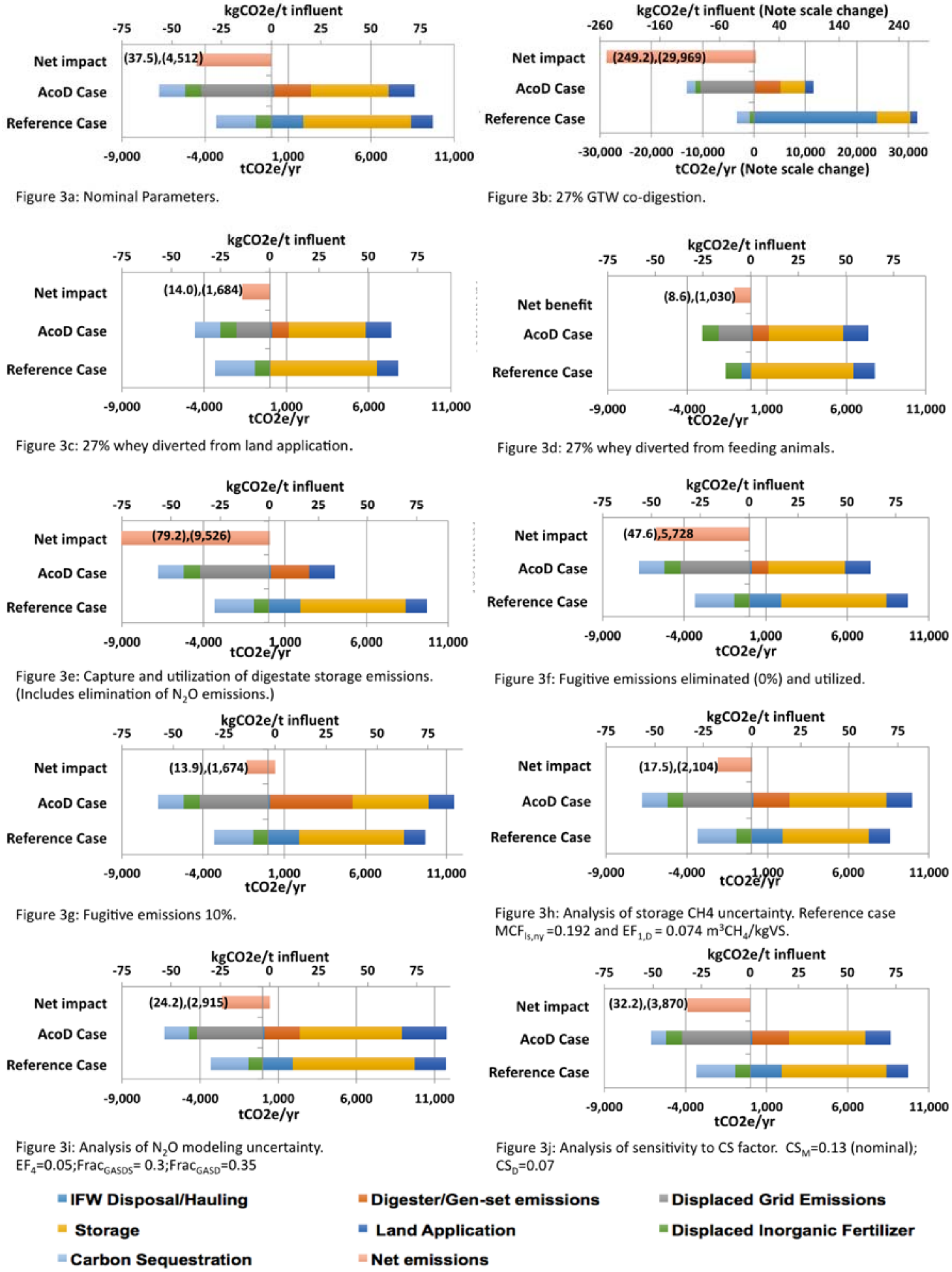


Figure 3a: Nominal Parameters.

Figure 3b: 27% GTW co-digestion.

Figure 3c: 27% whey diverted from land application.

Figure 3d: 27% whey diverted from feeding animals.

Figure 3e: Capture and utilization of digestate storage emissions. (Includes elimination of N_2O emissions.)

Figure 3f: Fugitive emissions eliminated (0%) and utilized.

Figure 3g: Fugitive emissions 10%.

Figure 3h: Analysis of storage CH4 uncertainty. Reference case $MCF_{ls,ny}$=0.192 and $EF_{1,D}$ = 0.074 m^3CH_4/kgVS.

Figure 3i: Analysis of N_2O modeling uncertainty. EF_4=0.05;$Frac_{GASDS}$= 0.3;$Frac_{GASD}$=0.35

Figure 3j: Analysis of sensitivity to CS factor. CS_M=0.13 (nominal); CS_D=0.07

Figure 3-3: Sensitivity analyses Comparison of AcoD, Reference case GHG emissions and net benefit in response to variation and uncertainty in parameters.

3.1.3.4 Nitrous emissions

N_2O emission estimates are subject to both uncertainty and variability. A simulation that varied all of the IPCC parameters related to land application N_2O emissions within their uncertainty ranges was used to analyze uncertainty (Appendix B, Table B-4). Climate change impacts from N_2O emissions varied by an order of magnitude (from 481 t CO_2e/yr to 6,476 t CO_2e/yr). The indirect emission factor (EF_4) for volatized N was found to have the largest impact (based on coefficient of correlation). However, varying EF_4 alone had little effect on net results because it was applied to both the reference and AcoD cases. The effect of a high indirect emission factor was greater when there is a difference between reference and AcoD case volatilization rates such as when the feedstock composition results in a TAN_D that differs significantly from TAN_M. In addition, variability in NH_3 emissions can arise from application technique and fertilization rates. Field experiments of manure application reported NH_3 emission varying from 2% of the TAN applied for slurry injection on arable land to 74% for broadcast surface spreading on grassland (Bartram and Barbour, 2004). Low emission techniques have the added benefit of preserving the amount of N remaining to displace inorganic fertilizer. Again, net impacts will be greatest when there is a difference between the reference and AcoD case application techniques. These uncertainty and variability impacts were explored through a sensitivity analysis where the volatilization rate for digestate was modeled to be higher than that of raw manure and a high indirect emission rate was assumed (Fig. 3-3i). The result was a 35% reduction in net benefit. Thus research or modeling to better understand indirect N_2O emissions can be important, especially in cases where elevated TAND is anticipated or when care is not taken to minimize NH3 volatilization.

3.1.3.5 Fertilizer displacement and carbon sequestration (CS)

Some studies have neglected fertilizer displacement, or when included have not considered inorganic fertilizer emissions (Borjesson and Burglund, 2006a, 2006b; Poeschl et al., 2012a, 2012b). Although this analysis includes the impact of fertilizer displacement, it is important to point out that the impact will only be realized if the nutrients are required by the system and do in fact replace inorganic fertilizer use. This is complicated by several factors, including the imbalance of nutrients in manure, difficulties in estimating nutrient availability, and low concentration of nutrients, making transport of organic fertilizer over long distances costly. It is unclear if a bias exists between the AcoD and reference cases. A sensitivity analysis assuming no change in fertilization practices in the AcoD case despite the import of food waste nutrients showed minimal impact (Appendix B, Fig. B-3). However, better understanding of true fertilization displacement due to AcoD may still be important, especially in cases where high nutrient content feedstock is imported or when large portions of the feedstock are not land applied in the reference case

CS has also been inconsistently applied in waste treatment LCAs, often neglected or narrowly analyzed. Long term studies of CS across different treatment pathways and for different substrates are lacking. In this study CS was consistently estimated across pathways as the long-term non-degradable fraction based upon substrate composition. A sensitivity analysis of a lower carbon sequestration factor for digestate (CSD=0.1) relative to manure (CSM=0.2) more than halved net benefit (Figure 3-3j). Thus research to better understand CS based upon waste composition for key waste treatment pathways may be important.

3.1.3.6 Other factors and study limitations

Operating parameters (i.e., HRT, OLR) and performance issues (mechanical or biological) can impact digester performance. The capacity factor (CF) is a measure of the performance of the digester system and is defined as the electrical energy generated by an engine gen-set relative to the maximum electrical energy that could have been generated in the same time period. It was calculated as 0.73 for this study. CF often improves over time; however, a study of 7 New York on-farm digesters reported an average CF of 0.57 (Gooch et al., 2011). Linear regression of a sensitivity analysis of CF (Appendix B Fig. B-4) resulted in a change in electricity generation of 110 MWh/percent CF and associated climate change impact of 59 t CO_2e/percent change in CF, for the NPCC regional grid mix.

While the impacts of several other parameters were explored (Appendix B, Figs. B-1 through B-4), it is not possible to generalize this study to all AcoD applications. This study analyzed climate change impacts of a state-of-the-art AcoD in Western New York, identifying key impacts and uncertainty. Furthermore, effort has been made to provide a clear methodology to be applied to other AcoD implementations.

3.1.4 Conclusions

A lifecycle analysis was performed on the basis of data from an on-farm AcoD in New York, resulting in a 71% reduction in climate change impacts, or net reduction of 37.5 kg CO_2e/t influent relative to conventional treatment of manure and food waste.

Displacement of grid electricity provided the largest reduction, followed by avoidance of alternative food waste disposal options and reduced impacts associated with storage of digestate vs undigested manure. These reductions offset digester emissions and the net increase in emissions associated with land application in the AcoD case relative to the reference case. Sensitivity analysis showed that using feedstock diverted from high impact disposal pathways, control of digester emissions, and managing digestate storage emissions were opportunities to improve the AcoD climate change benefits. Regional and parametrized emissions factors for the storage emissions and land application phases would reduce uncertainty.

3.2. *Life cycle greenhouse gas (GHG) emissions of a novel process for converting food waste to ethanol and co-products*

3.2.1 Introduction

Renewable transportation fuels have the potential to mitigate climate change and contribute toward energy independence and security. However, current fuels based on sugar or starch energy crops face significant challenges in terms of economics, availability of feedstock, land use conflict and life cycle greenhouse gas (GHG) emissions. Using waste as a feedstock offers an alternative that avoids many of these problems while also addressing the growing challenge of waste management.

Food scraps account for 21% of waste currently reaching landfills in the United States (U.S. EPA, 2010). In a landfill, food scraps decompose rapidly to produce methane, often before landfill gas (LFG) recovery systems are in place (Staley and Barlaz, 2009). Landfills accounted for approximately 16% of total U.S. anthropogenic methane emissions in 2010 (U.S. EPA, 2010). Alternatively, food waste can be broken down to simple carbohydrates and converted to ethanol in a bio-fermentation process. Using waste as a feedstock for ethanol production provides the service of waste disposal and has the potential to generate revenue to ethanol producers in the form of "tipping fees," which along with other valuable co-products can contribute to bio-refinery profitability.

Industrial (e.g. food processors) and retail (e.g. food preparation) wastes offer significant potential as a feedstock source because they can be source separated and are often a disposal burden to the generator. In particular, fruit juice and cannery waste have been reported as potential biofuel feedstocks (Fish et al., 2009; Nigam, 2000).

Food scraps, which are generally more complex lignocellulosic materials, also have the potential for conversion to ethanol. However, these substrates require the breakdown of starch, cellulosic or hemicellulosic materials into monomeric sugars to enable fermentation. One method of achieving this is simultaneous saccharification and fermentation (SSF) in which enzymatic hydrolysis is performed together with fermentation; this offers the benefit of reduced inhibition of enzymatic activity by saccharification end products, as well as reduced investment costs (Kumar et al., 2009; Olofsson et al., 2008). Although, empirical studies have demonstrated the potential to create ethanol from food scraps using SSF (Davis, 2008; Hong and Yoon, 2011; Kim et al., 2008; Ma et al, 2008), commercial-scale bioethanol plants utilizing food scraps do not yet exist. However, a National Renewable Energy Laboratory (NREL) model for lignocellulosic conversion based upon the SSF process has been used to analyze municipal solid waste (MSW) to ethanol conversion potential (Aden et al., 2002; Chester and Martin, 2009). Implementation of SSF can vary, but most processes are optimized to include an acid or thermal pretreatment and operate at elevated temperatures. Furthermore, commercial models are usually on the scale of 40–80 million gallons of ethanol/year and often include some form of cogeneration to utilize waste heat (Bellmar and Atieh, 2012).

Co-fermentation of feedstocks has received limited attention in the literature. Bellmer and Atieh (2012) and Dwidar et al. (2012) suggest that co-fermentation of beverage waste feedstock with other waste streams can improve pH, provide nutrients, and minimize diffusion of oxygen that might inhibit fermentation. Other studies have reported synergies when sugar- or starch-rich diluents were co-fermented with cellulosic feedstock (e.g., presaccharified wheat with wheat straw (Erdei et al., 2010) or furfural residue with corn kernels (Tang et al, 2011)).

This study analyzes a pilot fermentation plant where lignocellulosic food scraps are combined with a sugar rich diluent. The food scraps are ground without any other pretreatment and simultaneously co-fermented with diluent, at ambient temperature. The process produces ethanol as well as compost and animal feed co-products; the business model also encompasses revenue for the service of waste disposal. Furthermore, fermentation and dehydration are conducted at separate facilities. This distributed model minimizes the infrastructure and regulatory requirements at smaller fermentation facilities located close to waste streams, while taking advantage of economies of scale by conducting dehydration at a centralized hub.

The objective of this study is to estimate and analyze the climate change impacts of this novel process. Pilot plant (1/15th scale) fermentation data are combined with small-scale commercial distillation data to create a model of the full ethanol production process. This model is used to assess the life cycle climate change impacts and to evaluate the potential of the process as an alternative fuel pathway. The results are compared to those of corn ethanol and conventional gasoline. This study is unique in the literature in that it analyzes a process that produces ethanol from industrial food waste, whereas existing literature analyzes processes for the conversion of MSW to ethanol (Chester and Martin, 2009; Kalogo et al., 2007; Schmitt et al., 2012). Comparison of our results to these studies highlights the significant impact of waste feedstock composition which is discussed. Conclusions presented here are intended to contribute to knowledge in the areas of bioethanol production, waste management, and related policy.

3.2.2 Methods

3.2.2.1 Conversion Process Modeling

The process and system boundaries are shown in Figure 3-1. The bio-refinery process is modeled using primary data from the pilot fermentation plant and a commercial dehydration plant and supplemented with data from the literature (represented by shaded blocks) where primary data were not available.

A mass balance was performed for a control run at a pilot scale fermentation plant (10 wet t/day) operated by Epiphergy LLC. The control run consisted of 4.7 wet t of feedstock: 2.3 wet t lignocellulosic feedstock, consisting of food scrap waste from a supermarket chain and 2.4 wet t of diluted fruit syrup food processing waste as a diluent. The source-separated feedstock was transported from the waste generators in totes on trucks. Upon receipt, the food scraps were ground without any other pretreatment and mixed with the diluent. The mixture was combined with cellulose and starch biocatalysts and antimicrobial agents and simultaneously fermented with *S. cerevisiae* at ambient temperature. The resulting ferment slurry contained a dilute concentration of ethanol, residual solids, and yeast grown during fermentation. The solids were separated using an 80 um filterand fed into a composting process, which is accelerated by the grinding and fermentation. The volume and ethanol content of the filtered ferment, and mass of compost produced were measured. These processes are represented by steps1.1–1.4 of Fig. 3-5. In step 1.5 a portion of the dilute ferment is concentrated to create a Feed/Fuel Slurry (FFS) with 15% ABV. This is done to reduce transport weight as much as possible without requiring additional costs and regulatory burden associated with transport of flammable liquids. This process is modeled based on literature pertaining to small-scale ethanol distillation, assuming 0.22% ABV in the stillage (Stampe et al., 1983). Stillage wastewater volume, which is calculated by mass balance, was modeled to be processed onsite in a wastewater treatment (WWT) facility (Appendix B)

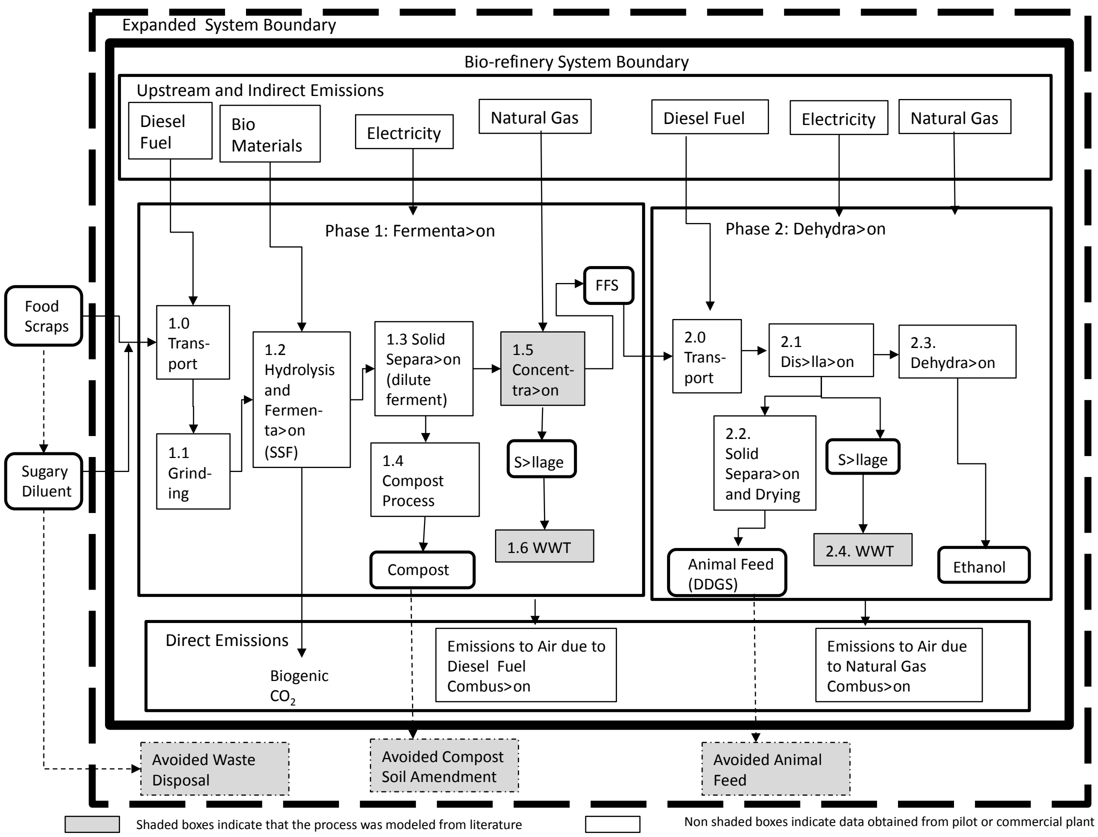


Figure 3-4: Ethanol production process and system boundaries. Bolded solid lines indicate the bio-refinery system boundary. Dashed bolded border indicates system expansion to net production process. Dashed arrows and processes indicate production processes for displaced co-products and services. Unit processes in gray are modeled based on the literature. Processes without a background are derived from pilot or commercial data.

The FFS is transported to a regional facility where it is distilled to 96.5% (ABV) and dehydrated using a molecular sieve to anhydrous ethanol. Dissolved solids and solids that were not removed by the filtering process at the fermentation plant, are separated and dried to create an animal feed product similar to dried distillers grains and solubles (DDGS). Wastewater is treated in an onsite WWT facility. The ethanol dehydration process is estimated to be 96.5% efficient.

3.2.2.2 Life Cycle Assessment (LCA) Methods

Goal and Scope

The objective of the analysis is to evaluate this waste-to-ethanol process as an alternative biofuel pathway in terms of global warming potential. A functional unit of 1 L of ethanol is used which is then converted to a unit of transport energy (1 MJ) for comparison to conventional gasoline (CG).

System Boundaries

The bio-refinery system boundaries are shown as bolded lines Fig. 3-5. It consists of two phases: fermentation and dehydration. The system boundary is set where the waste is introduced into the system. The food production processes that generate the waste are considered fixed with respect to process, materials, and consumption and thus not included within the boundaries (Friorksson et al., 2002; ISO, 2006).

The life cycle impacts include both indirect and direct emissions. These include, the indirect emissions associated with the production, transmission and distribution of electricity used in the process; the direct and upstream emissions from combustion of natural gas during phase 1 and phase 2 distillations; the production impacts of the material inputs to the process (biocatalysts and nutrients); and the life cycle emissions from diesel fuel required to transport waste material to phase 1 and FFS to phase 2. The impacts associated with the upstream production and construction of the phase 1 and phase 2 plants are not included in this analysis, as they are believed to be negligible per functional unit. Although this is supported by previous studies on corn ethanol, where they represent less than 1% of net GWP impacts (Farrell et al., 2006), verification in a mature, full-scale distributed ethanol system would be worthwhile in the future. Carbon dioxide created during the fermentation process is treated as biogenic and not included in GHG emission inventories (IPCC, 2006; U.S. EPA, 2006).

Treatment of Co-products

A variety of approaches exist for the treatment of co-products in LCA. In accordance with recommended guidelines (ISO14040/44) the system was expanded to model the displacement of competing products by the co-products generated in this process (ISO 2006).

The bio-refinery process produces two co-products, compost and an animal feed product (analogous to DDGS). Thus the net bio-refinery emissions account for the avoidance of the indirect and direct emissions associated with the production of these co-products through an alternative process. For the compost co-product this consists of the displacement of transportation and processing emissions associated with the alternative production of the compost. The resulting compost co-product is considered to be equivalent to compost produced by an alternative method and therefore the effects of compost application are considered equivalent and neglected in this analysis.

The treatment of impacts due to DDGS co-products in ethanol production has received much attention as it is shown to have a significant influence on results. A system expansion method is generally considered the most robust and most conservative method of treatment (Wang, 2005). Accordingly, the emissions are calculated for the nutritionally equivalent quantity of displaced animal feed. These include the indirect and direct emissions associated with the cultivation and production of displaced corn and soy meal. It also includes the net impacts on enteric fermentation due to the relative performance of feed DDGS relative to displaced corn and soy meal (Wang, 2012).

The service of waste disposal generates valuable revenue and is therefore also considered a co-product. Thus the net production emissions of the ethanol include the net bio-refinery emissions and the emissions due to avoided waste disposal. Because the waste feedstock used is diverted from the landfill, the system is expanded to include the avoided emissions associated with transportation of the waste to the landfill, processing of the waste at the landfill, and the net emissions associated with the decay of the waste at the landfill; these emissions are the sum of methane emissions released to the atmosphere, carbon storage within the landfill and avoided grid emissions due to methane captured by the landfill gas recovery system and used to create and displace grid emissions.

Life cycle inventory and impact assessment

Electricity, fuel and materials fluxes are compiled. Emission factors are applied to evaluate these fluxes for the midpoint impact category of global warming potential. The greenhouse gases considered are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). GHG emissions are aggregated on a carbon dioxide-equivalent (CO_2e) basis, using the 100-year global warming potential factors for methane and nitrous oxide emissions published by the Intergovernmental Panel on Climate Change (IPCC, 2012). These values are 1 for CO_2, 21 for methane, and 310 for nitrous oxide. Biogenic CO_2 produced in the fermentation process or any of the avoided waste disposal options analyzed is not included in the aggregated impact.

The lifecycle data sources and emission factors are summarized in Table 3-9.

Table 3-9: Life cycle data sources and emission factor sources

Impact	**Emission Factor**	**Source of parameters**	**Emission data source**
Transportation of feedstock	199 kg CO_2e/t-km	(t-km) calculated from pilot plant routes and payloads (t-km)	USLCI v1.6 database; transport, single unit truck, diesel, US, USLCI database
Transportation of FFS	199 kg CO_2e/t-km	(t-km) assumed to be 100km away, mass of FFS calculated from measured volume and density	USLCI v1.6 database; transport, single unit truck, diesel, US, USLCI database [1]
Electricity Consumption Phase 1	.771 kg CO_2e/kWh	(KWh) pilot plant data	SimaPro EcoInventv2.2 database; Electricity, medium voltage, at grid/US
Electricity Consumption Phase 2	.771 kg CO_2e/kWh	(KWh) commercial plant data	SimaPro EcoInvent v.2.2; Electricity, medium voltage, at grid/US
Natural Gas Consumption Phase 1	2400 kg CO_2/m3	(m^3) modeled from literature (m^3) (Stampe, 1983)	USLCI 1.6 database; Natural gas, combusted in average industrial boiler
Natural Gas Consumption Phase 2	2400 kg CO_2/m^3	(m^3) commercial plant data	SimaPro, USLCI 1.6 database; Natural gas, combusted in average industrial boiler
Biomaterial inputs	2776 kg CO_2e/kg	(kg and composition) pilot plant data data	Emission calculated as the sum of individual material inputs. Agonne National Lab.; GREET 1_2012rev2
Wastewater treatment at municipal WWT facility	0.518 kg CO2e/L ww	(L) calculated based upon pilot plant mass balance	SimaPro EcoInvent v.2.2; Treatment, potato starch production effluent to waste water treatment Class 2
Wastewater treatment onsite POTW	.003kg CO_2e/L EtOH	based upon Corn Ethanol (Farrel, 2006)	Farrel, 2006; EBAMM v. 1.1
Avoided Animal Feed	-167 kg CO2e /dry kg	(kg) DDGS calculated based upon TS measurement in FFS	Farrel, 2006; EBAMM v. 1.1.
Avoided Compost	-92.59 kg CO_2e /kg	(kg) pilot plant data	EPA,2012; WARM v.12
Avoided Landfill	-2535 kg CO_2e /dry t	(dry t) calculated based pilot data	EPA,2012; WARM v.12 - adjusted to per dry metric ton

Table 3-10: Deliveries of food waste feedstock to the pilot. Calculation of t-km travelled

	(kg) loaded	Cumulative (kg) transported	km to next stop	t-km
Location 1	377	377	20	7.5
Location 2	343	720	6.5	4.7
Location 3	374	1094	6.4	7.0
Route 1- day 1	1094	1094	32.9	19.2
Location 1	267	267	49.2	13.1
Location 2	24	291	11.3	3.3
Location 3	0	291	17.5	5.1
Route 2- day 1	291	323	78	21.5
Route 3-day 1	1000	1000	26.2	26.2
Location 1	355	355	20	7.1
Location 2	50	405	6.5	2.6
Location 3	185	590	6.4	3.8
Route 2- day 1	590	235	32.9	13.5
Location 1	398	398	49.2	19.6
Location 2	27	425	11.3	4.8
Location 3	140	565	17.5	9.9
Route 2-day 2	565	565	78	34.3
Route 3-day 2	1400	1400	26.2	36.7
total t-km	151.4			
total kg transported *	4649			
total km travelled	274.2			
average payload (t)	0.55			

* Amount transported does not equal amount loaded into the processes exactly due to stocks and flows in the feed system

Electricity consumption at phase 1 was estimated based on an inventory of equipment (grinders, augers, pumps and separators), rated or measured current draw, and time of use measurements (Table 3-11).

Table 3-11: Phase 1 pilot process Energy Analysis Summary

Sub Process	Rated Capacity kW-hr	Baseline Measured kW-hr	Sensitivity A kW-hr †	Sensitivity B kW-hr ††
Bulk Liquids	0.5	0.4	0.4	0.2
Grinding	276.5	176.6	157.3	157.0
Primary Fermentation	52.8	48.2	39.6	26.4
Secondary Fermentation	1.4	0.7	0.7	0.7
Solid/Liquid Separation	140.7	63.7	62.3	59.7
Destruction	0.7	0.7	0.5	0.4
Distillation †††	2.32	1.34	1.29	1.19
Composting	3	2.1	2.0	1.6
Total				
Total KWH/Process	**478.0**	**293.8**	**264.1**	**247.2**
Total MBTU/Process †††	1.63	1.00	0.90	0.84

†Assumed that the regrind pump current draw is proportional to the grinding motor current draw (M3), and pumps running at 75% Duty, Based on M2 and M8 in Process flow diagram

††Assumed that the regrind pump current draw is proportional to the grinding motor current draw (M3), and pumps running at 50% Duty, Based on M13 and M17 in Process flow diagram

†††Does not include Natural Gas use to run still boiler

Emissions for electricity consumption were based on the U.S. Average Grid Mix, using the Ecoinvent v2.2 emission factor for electricity, medium voltage, U.S. (EcoInvent Centre, 2007). Specific biomaterial inputs and quantities were provided by Epiphergy LLC and are considered proprietary. However, they were used to calculate the life cycle emissions for biomaterials using factors obtained from the GREET model (Wang, 2012) and the EcoInvent 2.2 database (EcoInvent Centre, 2007) for the individual biomaterials. Phase 1 natural gas consumption was calculated for the concentration of the dilute ferment using 5 MJ/L anhydrous ethanol (Stampe et al., 1983). Natural gas emissions from concentration were calculated to account for provision and combustion of the natural gas in an average industrial boiler operating at 85% efficiency using USLCI v1.6 life cycle emissions data (USLCI, 2012) Wastewater treatment at phase 1 is assumed to be performed at an onsite wastewater treatment plant (Farrell et al., 2006). An emission factor from Ecoinvent v.2.2 for the treatment of potato starch effluent at a class 2 wastewater treatment plant is used (EcoInvent Centre, 2007).

Since a phase 2 plant does not exist at this time it was assumed to be 100 km away from the phase 1 plant with FFS transported by a single-unit diesel-operated truck. Data on natural gas and electricity consumption on a per liter basis to dehydrate FFS to anhydrous ethanol as well as evaporation energy to produce the animal feed product were provided by Merrick and Company which has been operating a similar plant since 1996 (Table 3-12) (Wagner, 2013). This plant processes brewery waste to ethanol and produces 3 M gallons of ethanol per year. The emissions associated with onsite processing of wastewater generated at the phase 2 distillation plant were based upon the emission factor per unit wastewater treated at a corn ethanol plant (Farrell et al., 2006).

Table 3-12: Small Scale Biorefinery Data

Natural gas/ gallon EtOH	8.44 MJ (8000 Btu)	Based upon 5% ABV FFSinput to distillation
Electricity/gallon EtOH	1.4KWh	
Natural gas/gallon stillage	7.49 (7100 Btu)	Based upon 15% TSS to dry to 90% DMB

Source: Steve Wagner, Merrick and Company, 2012

Displaced landfill emissions are based on the EPA WARM model using the category of 'food scraps' (U.S. EPA, 2012a; U.S. EPA, 2012b). An adjustment is made based on the dry mass to determine an equivalent mass of food scraps avoided at 70% moisture content. Additionally, a component-specific decay "k-constant" of 0.08 was used representing wet landfill conditions to account for additional moisture. The landfill is modeled to have the current national average LFG recovery system and national average electricity grid mix.

The U.S. EPA WARM model was also used to calculate the avoided transportation and processing emissions related to the compost co-product. These are calculated based on the equivalent mass of feedstock required to create compost by an alternative method, using the conversion of 2.1 t of yard waste to create 1 t of compost (U.S. EPA, 2012a).

The displacement credit for the animal feed co-product is calculated based on the DDGS displacement from the EBAMM v.1.1 model (Farrell et al., 2006; Grabowski, 2002). The GWP impact is scaled to account for the quantity of feed co-product produced.

3.2.3 Results and Discussion

3.2.3.1 System Modeling Results

The process modeled here produced 276 liters of anhydrous ethanol, 160 kg of compost and 428 kg of feed from 4.7 wet t of wet industrial/retail food waste equivalent to of 296 L EtOH/dry t feedstock (or 0.23g EtOH/g dry solids)(Table 3-13). The theoretical yield for this process is estimated to be 585 L EtOH/dry t of feedstock (calculation in Appendix B, Equation 3). Therefore the co-fermentation process efficiency is estimated to be 54% of theoretical yield.

Table 3-13: Summary of process inputs, outputs and yields resulting from pilot plant audit and mass balance.

Process Inputs	
Organic Waste Input (wet mass) (kg)	**4718**
Food Scraps (at 70% moisture) (kg)	2309
Syrup diluent (at 90% moisture) (kg)	2409
Biomaterial inputs (kg)	**9**
Estimated Total Feedstock Dry Mass In (kg)[a]	*934*
Process Outputs	
Phase 1: Ferment (5.88% ABV) (liters)	**4978**
Phase 1 estimate: FFS @15% ABV with 0.22% ABV in stillage (liters)	*1904*
Phase 2 estimate: Anhydrous ethanol yielded calculated from FFS at 15% ABV assuming 96.5% efficiency (liters) [b]	*276*
Phase 2 estimate: Animal Feed calculated based by mass closure (kg) [c]	*428*
Phase 2 estimate: Stillage calculated from mass balance of Phase 2 distillation (liters)	*1628*
Phase 1 estimated Stillage based upon mass balance of distillation phase (liters)	*3406*
Phase 1: Compost at 50% moisture (kg)	**160**
Yield liters EtOH/ dry t	295
Estimated Theoretical Yield (liters / dry t)	544
% Theoretical Yield	54%

[a] Does not include recycled process liquid or biomaterial inputs

[b] Calculated from ethanol balance (supplemental materials)

[c] Dry Mass In=DDGS+EtOH mass+ CO_2mass loss+compost solids

Previous studies on the conversion of organic MSW to ethanol have produced a broad range of results. Kalogo et al. (2007) reported approximately 121 L EtOH/dry t (85 L/ wet t) using u an acid hydrolysis Gravity Pressure Vessel pilot process using MSW fluff (consisting of food, paper, and yard waste). In contrast, Schmitt et al. (2012) modeled a process based on lab experiments using dilute acid hydrolysis pretreatment followed by SSF at 30°C on a synthetic MSW feedstock (defined as banana peels, cereal, coffee grounds, canned corn, tomato juice and clean hygiene products) and reported a process yield of 469 L/dry t and 74% of theoretical yield. Thus the yield reported here is within the range of reported values on a mass basis. However, it is worth noting that the composition of the co-fermentation substrate has a lower lignocellulosic content than MSW due to the contribution of the sugary diluent, which we would expect to have higher conversion efficiency. Furthermore, increased conversion of lignocellulosic material is likely to require more inputs and increase production costs. This process differs from other published methods for lignocellulosic SSF in that it functions at lower operating temperatures (20°C vs. 37°C) and involves minimal pretreatment. Because the bio-refinery generates revenue from compost, animal feed and waste disposal (tipping fees) as well as ethanol, it is unclear if maximizing ethanol yield would necessarily maximize profits.

3.2.3.2 LCA Results and Analysis

<u>Comparison to corn ethanol and gasoline</u>

The GWP impacts for the process are compared to those of corn ethanol production using a functional unit of 1 liter of ethanol. In order to compare the results to conventional gasoline (CG) they are converted to a MJ basis to account for the difference in performance between ethanol and gasoline. (Table 3-14). These results show a net carbon-negative production process with 553% improvement in GWP impacts relative to corn ethanol and 460% relative to conventional gasoline. This reduction is almost entirely due to the avoided methane emissions that would be incurred by food waste disposal in a landfill. Without the inclusion of avoided landfill impacts, the net bio-refinery emissions (phase 1 and phase 2) show a 9% improvement over commercial corn ethanol production (including agricultural phase impacts).

Table 3-14: Life Cycle GWP impact results (gCO_2e/L EtOH) and comparison to corn ethanol and gasoline (gCO_2e/MJ)

	This Study	Corn ethanol[a]
Total Bio-refinery Emissions (gCO_2e/L)	1458	
Displaced Landfill emissions	-8590	
Net Bio-refinery emissions (gCO_2e/FU)	**(7132)**	**1608**
Reported HV of Ethanol (MJ/l)	21	21
Net Production Process (gCO_2e/MJ)	**(340)**	77
Ethanol distribution (g CO_2e/ MJ)	1	1
Net Produced and distributed (gCO_2e/MJ)	**(338)**	**77**
Difference between corn EtOH (g CO_2e /MJ)	416	0
% Difference improvement between corn EtoH	**-554%**	**0%**
gCO_2e per MJ of Conventional Gasoline (CG) produced, distributed, and combusted.	**94**	94
Percentage difference to CG	**-460%**	**-17%**

[a] Farrel et. al, 2006

Contributional Analysis

Table 3-15 shows the life cycle contributions of the two production phases and landfill avoidance to total process emissions and compares it to corn ethanol emissions. Phase 1 has a larger contribution (1217g CO_2e/L EtOH) to emissions than phase 2 (241g CO_2e/L EtOH). This is driven by electricity use (816 g CO_2e/L EtOH) to operate grinding and separating equipment followed by natural gas use for concentration (241g CO_2e/L EtOH). Compost co-product production provides a small offsetting credit (54g CO_2e/LEtOH). Phase 2 accounted for only about 22% of process emissions. Electricity use (285g CO_2e/L EtOH) is again a major driver followed by natural gas for distillation and drying (146g CO_2e/L EtOH). However, the large credit for animal feed production (260g CO_2e/L EtOH) reduced emissions for this phase by nearly 50%. The large contribution of electricity and natural gas consumption to process phase emissions indicates that cogeneration of electricity or heat, as is common in cellulosic ethanol processes, may be an opportunity. The life cycle emissions associated with enzymes and other biomaterial inputs (91g CO_2e/L EtOH) is relatively small due to the small amount of biomaterials used.

Table 3-15: Contributional analysis of life cycle GWP impacts (gCO2e/L EtOH)

This study	**gCO_2e/ L EtOH**
Phase 1 emissions	1,271
Transportation of waste	109
Electricity consumption	816
Natural gas consumption	249
Biomaterial inputs	91
POTW	6
Avoided compost co product	(54)
Net emmissions Phase 1	**1,217**
Phase 2 emissions	501
Transportation of FFS	66
Electricity Consumption	285
Natural Gas Consuption	146
WWTF	3
Avoided animal feed co product	(260)
Net emissions Phase 2	**241**
Net biorefinery emissions	**1,458**
Displaced Landfill emissions	(8,590)
Net production emissions	**(7,132)**

The impact of transportation in this two-phase model was also analyzed. It is found to make a noticeable contribution, representing approximately 12% of the process emissions (phase 1 and phase 2 emissions only). Transportation of the FFS to phase 2 (66g CO_2e/L EtOH) has only 40% of the emissions impact of transportation of feedstock to the phase 1 plant (109g CO_2eL EtOH). This is due to the reduction in mass transported due to the removal of moisture and solids as a result of the fermentation, separation and concentration processes at phase 1. Furthermore, when avoided waste disposal emissions are considered, the impact of feedstock transport is more than offset by the avoided transportation of waste feedstock to a landfill (calculated using the WARM model). This result is attributed to the lower impact of single unit trucks with only a few collection stops, as compared to modeling heavy waste collection vehicles that make frequent stops. Nonetheless, transportation and in turn the location of the phase 1 and phase 2 plants do impact GHG emissions and will require optimization with process scale-up.

Comparison to other waste-to-ethanol LCAs

Comparisons of life cycle results to other waste-to-ethanol processes are challenging and considerations have been made to provide a meaningful comparison. First, the phase 1 emissions used in this analysis are based on a pilot plant facility operating at 1/15th its intended capacity. Thus it is considered a worst-case scenario since a full-scale production facility will likely benefit from learning curve and scale economy effects. Additionally, treatment of avoided waste disposal in the previous LCAs was inconsistent and highly influenced by feedstock (Chester and Martin, 2009; Kalogo et al., 2007; Schmitt et al., 2012). Therefore, results for just the production process are compared first and a discussion of treatment of feedstock is presented in the next section.

Lastly, differences in processes, co-products, and analysis objectives have also been considered in comparing results. Kalogo et al. (2007) reported on MSW-to-ethanol using a dilute acid hydrolysis and Gravity Pressure Vessel technology. Their results show that the classification process to remove inorganic material) has a large contribution to emissions (nearly 40%); results are therefore shown with and without classification. Schmitt et al. (2012) used the NREL ASPEN model of a dilute acid SSF process on lignocellulosic material. In this system, residual lignin is combusted to generate electricity, offsetting site usage. Chester and Martin (2009) perform an Economic Input/Output LCA also using the NREL model, with the objective of comparing the business-as-usual MSW system in California to one of waste-to-ethanol. They do not include waste collection as they rationalize that it would occur in either case.

Despite significant differences in scale and implementation, the results of this process fall within the range of the other studies (Fig. 3-5). Thus it may be concluded that the impacts of smaller scale and process optimization tradeoffs are offset by less process inputs and the selection of highly degradable, source separated feedstocks.

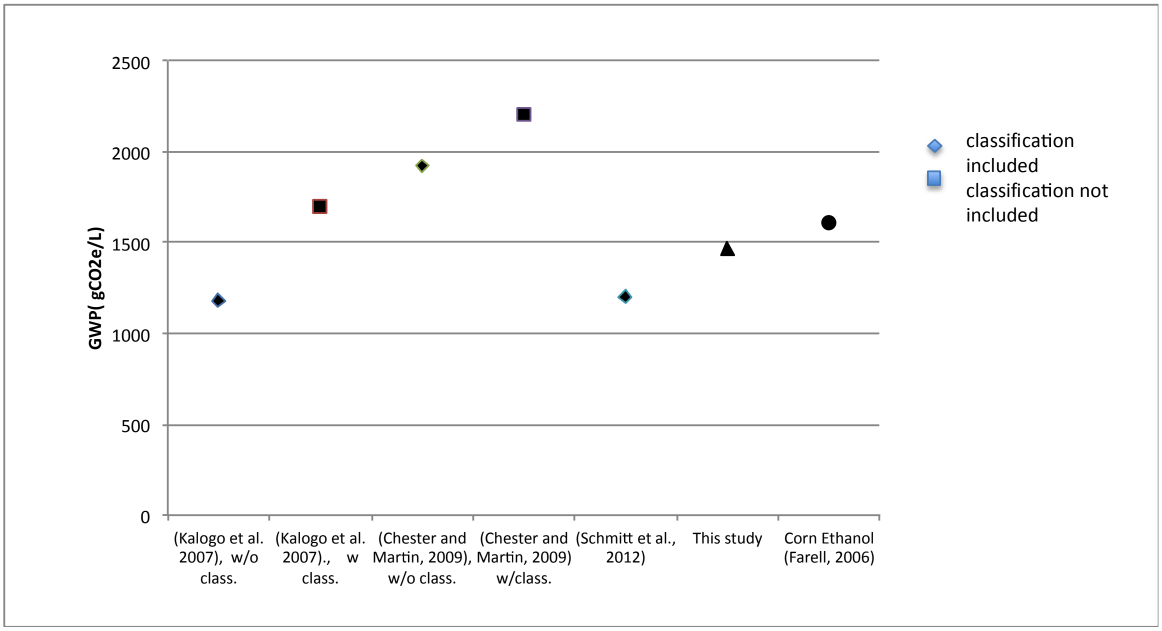


Figure 3-5: Comparison of net bio-refinery process life cycle GWP impact results (gCO_2e /L EtOH). This includes the process itself and does not include avoided waste disposal of feedstock or ethanol distribution. This study is compared to MSW to ethanol studies and corn ethanol.

Food waste vs. MSW

The studies discussed in the previous section have all utilized organics derived from MSW where this study utilizes industrial and retail food waste (Chester and Martin, 2009; Kalogo et al., 2007; Schmitt et al., 2012). In addition to the reduction in classification required due to source-separated feedstock, the characteristics of the feedstock can have a significant effect on life cycle GWP results. LFG emissions are a function of the rate of decay of the waste and the potential of the waste to generate methane. Food scraps have a high potential for methane generation as well as a rapid decay rate. Due to the phased implementation of typical LFG recovery systems, rapid decay of food scraps results in net GHG emissions, even with aggressive LFG recovery

(U.S. EPA, 2012a; U.S. EPA, 2012b). (Uncaptured methane is indicated by the area between the solid lines and the dashed lines in Fig. 3-6.)

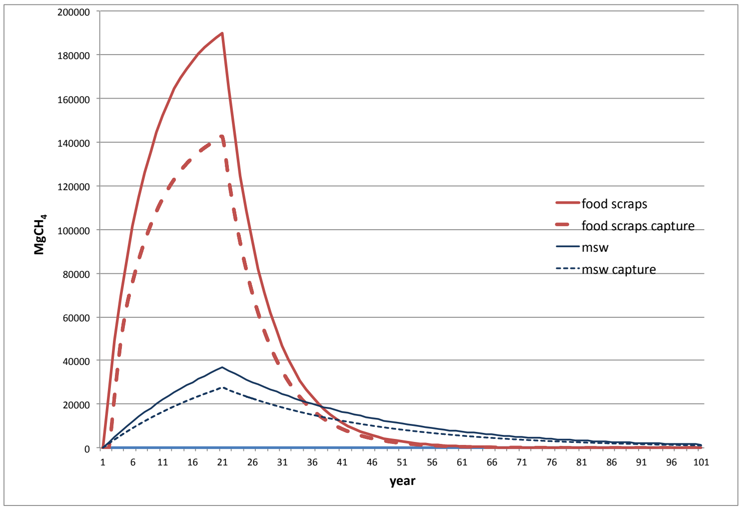


Figure 3-6: Methane production rate (m^3 CH_4/year). Comparison of methane production (solid lines) and LFG captured (dashed line) for MSW and food scraps over 100 years. Calculated using LandGEM v3.02, based on 1 t of waste. For Food Scraps: k = 0.14, L_o = 301; MSW: k = 0.04, L_o = 100. Phased-in methane collection: Years 1–2: 0%, Year 3: 50%, Year 4: 70%, Years 5–100: 75% (U.S. EPA, 2005;De la Cruz and Barlaz, 2010)

In contrast, when Kalogo et al. (2007) considered MSW derived organics containing yard waste and paper waste as well as food scraps and Chester and Martin (2009) considering MSW including construction /demolition and paper waste, both found that the net avoided GWP emissions flipped from positive with no LFG recovery system to negative with LFG recovery to electricity generation. Food scrap diversion from a landfill always results in positive avoided emissions regardless of the LFG recovery system reducing the sensitivity of the net impacts to landfill technology.

Sensitivity to avoided waste disposal

While this process utilized feedstock that was diverted from the landfill, alternative disposal pathways were also analyzed to determine the impact on our results. For solid waste the alternatives of landfill or industrial composting were modeled. For the diluent, the alternatives of wastewater treatment and land application were modeled in addition to the base case of avoided landfill. Avoided waste disposal emissions are affected by waste composition, technology, environmental conditions and modeling methodology. Data on waste disposal alternatives is limited and emissions factors based upon specific food waste characteristics were not available. In most cases the general category of food waste is modeled. Four scenarios were analyzed and compared to the base case of feedstock diverted from landfill and the net emissions of corn ethanol (Fig. 3-7). Error bars indicate the range of values related to technology and environmental conditions.

Solid waste disposal influenced results more than liquid disposal options. Emissions due to landfilling of waste has the largest magnitude of impact, ranging from 1576 kgCO_2e/wet t food scraps with no LFG recovery to 375 kgCO_2e/wet t food scraps for LFG recovery to electricity. Nevertheless, the net result of diversion of food scraps from a landfill is a significant avoidance in emissions. Thus the net ethanol process remains carbon-negative for all scenarios with landfill avoidance.

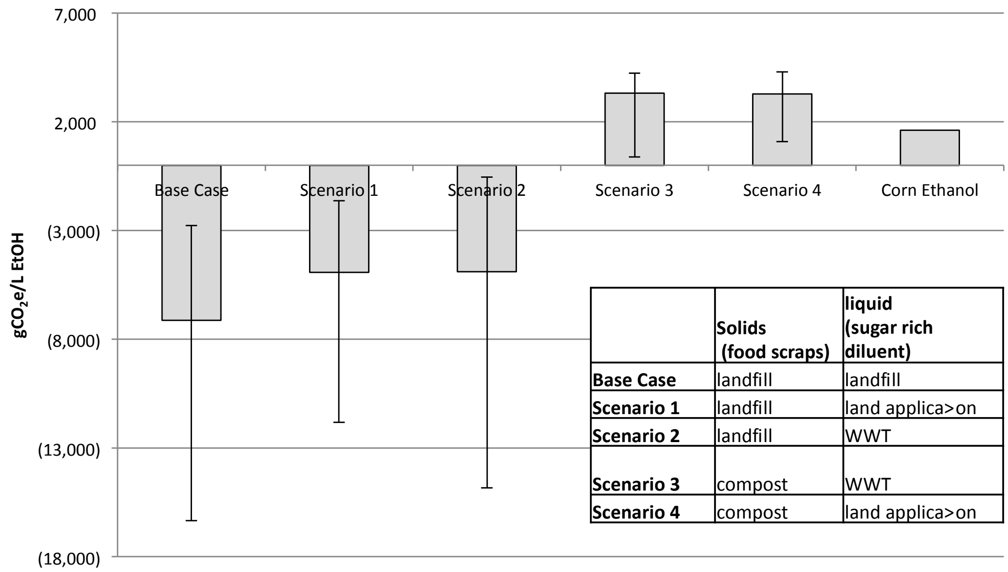


Figure 3-7: Sensitivity of results to avoided waste disposal treatment (g CO_2e/L EtOH). Four scenarios are compared to the base case and to corn ethanol. The base case represents the case reported in this study, where all waste feedstock is diverted from the landfill. The four scenarios consist of either landfill or composting of solids and either wastewater treatment or land application of liquid feedstock and are shown in the inserted table. Error bars indicate the range of results due to technology and environmental conditions.

When considering diversion of waste from a commercial compost facility, soil carbon storage resulting from the application of the compost is considered along with the emissions incurred due to transportation and processing of the waste per the EPA WARM model (U.S. EPA, 2012). The net application of compost results in the sequestration of -220 kgCO_2e/t waste processed. Therefore diverting waste from the carbon-negative compost process increases net GWP emissions for the waste-to-ethanol process. Scenarios that utilized solid waste diverted from composting resulted in higher net emissions than corn ethanol. However, these results were sensitive to uncertainty in the amount of carbon storage as well as direct CH_4 and N_2O emissions due to the compost process. The amount of carbon storage is affected by soil characteristics and application schedule. A best-case carbon storage scenario, resulting in net sequestration of -331 $kgCO_2e/kg$ waste is shown through the lower end of the error bars. Research into CH_4 and N_2O emissions from composting is ongoing and not yet included in the WARM model. While these emissions are considered in the IPCC methodology they are quite small ($4gCH_4/kg$ wet waste processed) and ($0.3gN_2O/kg$ wet waste processed) (IPCC, 2006). A worst case is constructed to include the direct CH_4 and N_2O emissions along with emissions due to fossil fuel used to process the compost, but not including any carbon storage effect. Since in this case, diversion of food waste from the compost process would eliminate these emissions, this represents the lower end of the range shown by the error bars. (Displacement of fossil based fertilizer is not considered in this analysis but could also influence results). Finally, this study does not take into consideration any difference due to application of compost derived from food waste (as in this study) and compost derived from yard waste as in typical of some commercial compost processes (U.S. EPA, 2012).

There is limited U.S. emissions data on wastewater treatment and land application of food processing wastewater. Similar to solid waste, treatment technology, waste characteristics and modeling affect uncertainty in wastewater emissions. A baseline wastewater treatment emission factor of $1.3kgCO_2e/m^3$ wastewater (ww) based upon fruit and vegetable processing wastewater was used (U.S. EPA 2013). However, this process only includes direct CH_4 and N_2O emissions and does not include fossil fuel use or infrastructure, which some studies consider quite large. Therefore data from a range of relevant processes in EcoInvent v2.2 were used to model uncertainty ranging from 0.51 kgCO_2e/m3 for potato starch wastewater treatment to 83.3kg CO_2e/m^3 ww for treatment of organic wastewater (EcoInvent, 2008). In all cases wastewater processing results in net emissions, although in some cases quite small, thus avoidance of this process contributes to emission savings for the ethanol production process.

Land application emission factors for food waste were not available. Net emissions due to land application are the result of CO_2 emissions due to spreader fuel consumption as well as the net impact of CH_4 and N_2O emissions, carbon storage and fertilizer displacement. Similar to composting, the latter impacts are influenced by the soil conditions, waste characteristics and agricultural practices. Land application emissions were modeled to range from 1.21kgCO_2e/m^3 ww due to avoidance of emissions to operate the spreading equipment only to -8.5kgCO_2e/ kg ww based on studies of manure spreading net impacts (EcoInvent, 2007; Moller et al., 2009). Thus diverting waste from compost and land application to the ethanol production process studied herein has the least potential for GHG reduction.

Waste disposal alternatives are driven by many factors including economics, waste characteristics (i.e., solid, liquid, packaged, etc.) as well as market availability. Furthermore, they may vary by type of waste, region and over time for a given waste to energy process. Our results indicate that it is important to understand and model the appropriate waste disposal scenarios to understand the net impact of waste to energy processes.

3.2.4 Conclusions

Process yields for co-fermentation of lignocellulosic material with sugar-rich diluent, using SSF with a grinding pretreatment, based upon pilot plant data are reported. Life cycle GWP impactsfor the process are comparable to commercial processes studied in the literature. Furthermore when the avoidance of landfill emissions is considered, the process shows a significant improvement over corn ethanol or conventional gasoline with respect to GWP impacts. The results indicate that the use of readily convertible, source-separated commercial or industrial food waste as a feedstock for ethanol offers significant potential for GHG reduction. Furthermore, important to understanding the life cycle impacts of corn ethanol, this study illustrates how feedstock and alternative waste disposal options have important implications in life cycle GHG results for waste-to-energy pathways.

3.3. *Conclusions*

Several conclusions were drawn across both emerging technologies analyzed. Both technologies were promising in terms of GHG reduction based upon specific systems studied. The impacts of implementing these technologies are inherently comparative, meaning that they must be interpreted in the context of the alternative utilization of the waste feedstock as treatment of waste is required in any event. Furthermore, the avoided treatment pathway can have a significant influence on net results. This observation motivated work in Chapter 4 to understand the impact of alternative utilization pathways. Treatment of co-products and co-services (i.e., compost, electricity, fertilizer) can also be significant. Thus further research into waste derived biofuel lifecycle impact assessment is not only suggested but it is recommended that it be coordinated to ensure consistency and comparability across studies. Finally, the climate change impacts associated with transportation of industrial or commercial FSC resources were generally small indicating that it may not warrant the significant attention often given in LCAs. Although it is worth noting that collection and transportation can be more significant in residential applications. Furthermore, although climate change impacts were small the financial impact of transportation is often a significant consideration.

Chapter 4 Evaluation of anaerobic digestion of commercial food waste and co-digestion with manure: characterizing biochemical parameters and synergistic effects

4.1. *Introduction*

As presented in Chapter 2, commercial resources (generated in the retail and food service sector) were estimated to constitute about 40% New York State MSWFW. Furthermore, 97% of commercial resources were estimated to be landfilled. Chapter 3 noted the high impact of food waste landfilling and the potential to deduce climate change impact through anaerobic co-digestion.

This chapter discusses experiments conducted to characterize several types of resources generated by the commercial sector of the New York State food supply chain. It begins with an Introduction containing the motivation and objectives of this work, followed by a Methods section which explains feedstock selection and preparation as well as the experiments performed. The Results section presents data on the characteristics of the FSC resources, as well as bio-degradability parameters and co-digestion performance data.

Anaerobic digestion (AD) has been promoted for its ability to generate clean renewable energy, treat waste and recycle nutrients. Early adoption of AD in the U.S. has primarily occurred on concentrated animal feedlot operations (CAFOs) where it also provides odor reduction and increased manure management flexibility. The number of on-farm anaerobic digesters in NYS has grown to 33 (Chapter 2) from 3 in 2002 (Agstar, 2002). However, single substrate digestion (mono-digestion) of manure can result in low biogas yield due to low organic load and high N concentrations of manure may lead to inhibition and process instability. Combining feedstock substrates or anaerobic co-digestion (AcoD) can increase organic loading and improve performance relative to mono-digestion by diluting toxic or inhibitory compounds and providing macro or micro nutrients (Khalid et al., 2011; Mata-Alvarez et al., 2011). In addition, AcoD of manure and food waste can improve project economics through additional revenue in the form of "tipping fees" for the imported food waste. Thus recent years have witnessed a trend toward AcoD with 98 of the 260 farm-based biogas plants in the U.S. now co-digesting additional feedstocks (U.S. EPA, 2015). This trend is consistent with observations reported in Chapter 2, where 7 on-farm digesters in NYS reported co-digestion (with13 permitted or registered to do so but not yet reporting volumes) vs. zero in 2002. With this trend has come the need to develop methods that could improve the performance as well as the efficiency of this process, including analysis of co-digestion substrates to exploit their complementary characteristics and the use of mathematical models simulating the AcoD process, as recognized by Esposito et al. (2012).

Currently, industrial food processing wastes and agricultural wastes are the predominate co-digestion feedstocks (U.S. EPA, 2015). However, increasing regulation of organic disposal in landfills is driving interest in AcoD among solid waste generators (Massachusetts, 2013). These landfill bans or mandates often target commercial establishments that landfill large quantities of food waste. Commercial food waste is mainly composed of retail food waste and food service waste. Supermarkets are a large source of retail food waste where prepared foods, supply in excess of demand or non-conforming products results in scrap, rotting produce perishables, damaged packaged goods or otherwise unmarketable product. Food service waste consists of scraps generated during food preparation as well as post-consumer plate waste and un-served food. While a portion of commercial food waste may be reduced or diverted to feed the hungry, some commercial food waste is unavoidable. In fact, over 40% of the food produced in the U.S. ends up in a landfill without reaching a table, from which 19% originates from the retail-level food supply (Gunders, 2012).

Commercial food waste generated from different operations within an establishment or at different types of establishments can be categorized and often source separated. These waste products can become valuable resources for renewable energy production when anaerobically digested or co-digested. While AcoD has received increasing attention in the literature, most studies have focused on the organic fraction of municipal solid waste (OFMSW), industrial wastes or agricultural wastes as co-digestion feedstocks (Mata-Alvarez et al., 2011). This study has collected a representative array of commercial organic waste substrates to analyze as feedstock for AD. The objectives of this research were threefold: 1) provide data on representative commercial food waste composition; 2) provide key biodegradability parameters, namely bio-methane potential, degradation extent and hydrolysis rate coefficients; and 3) assess potential synergistic or antagonistic effects when these complex substrates are co-digested.

4.2. *Methods*

4.2.1 Substrate description

<u>Source-separated commercial food wastes</u>

Samples of retail food waste were obtained from the food bank for the Finger Lakes region of New York (Foodlink, Inc.) where non-distributable food was source-separated into several retail waste categories: fruit and vegetable waste (FVW), stale baked goods (BG), damaged canned goods (CG), non-distributable yogurts and frozen desserts (YFD), salad mix waste (SM), and dried goods, which were further separated into sweet (SDG) and unsweetened (UDG) dried goods (Table 4-1). Further, the following kitchen waste samples were obtained from the source-separated waste collection bins of the Grace Watson dining hall (GWDH) at the Rochester Institute of Technology: kitchen food preparation waste (PREP), cafeteria spent coffee grounds and filter paper (COF), and post-consumer waste (POST) and soiled napkins (SN) from the returned trays after meals (see Ebner et al., 2014b). Approximately 20kg of each of the 11 substrate samples were collected. Samples were stored at 4ºC until prepared (approximately 5 days) and then immediately frozen until used again. The substrates were first manually mixed, and then ground, using a VitaMix® blender (1825 Professional Series 750) to reduce particle size to less than 2mm and produce a homogenous slurry or powder material.

Food sector co-digestion blends

Selected source-separated food wastes were combined to model the potential co-digestion waste stream originated at three commercial food sectors: 1) Cafe (CAFE) –

combined BG and COF in a 60:40 proportion (% fresh weight (%w/w)) 2) food service waste (SERVICE) – combined POST and PREP in a 80:20 proportion (% w/w) (see Ebner et al., 2014); and 3) retail (RETAIL) – combined FVW (57%), SDG (7%), BG (21%), CG (8%), YFD (7%) (% w/w) to replicate the reported composition of the food bank waste.

Manure-food waste co-digestion blends

Dairy manure slurry (M) was co-digested with food wastes and sector blends in a 70:30 ratio (%w/w) chosen based upon data reported on New York State's largest manure-based anaerobic co-digestion facility (Ebner et al., 2015b). The dairy manure slurry was obtained from the receiving pit of a dairy farm equipped with a scrape manure collection system. The 24 substrates evaluated are summarized in Tables 4-1.

Table 4-1: Description and sources of substrates evaluated

Source separated commercial food waste		
Substrate	Description	Source
Baked goods (BG)	Stale bagels, muffins and donuts.	Foodlink
Canned goods (CG)	Damaged cans of crushed tomatoes, diced tomatoes, green beans, beets, chicken broccoli soup, cream of chicken soup, cheese pot pie soup, baked beans, papaya, pineapple chunks, tuna fish and mandarin oranges in damaged plastic cups	Foodlink
Coffee grounds (COF)	Spent coffee grounds (medium roast) and coffee filter paper	GWDH
Sweet dry goods (SDG)	Assorted breakfast cereals (Cocoa O's©, Cap'n Crunch©, Shredded Wheat©, Lucky Charms©, Chex©, Frosted Flakes©) and dry goods (quick oats, pasta, Cliff© cereal bar)	Foodlink
Unsweetened dry goods (UDG)	Assorted grains (rice, oatmeal, bread crumbs, cream of wheat)	Foodlink
Fruit and vegetable waste (FVW)	Approximately 50% rotting bagged lettuce and 50% rotting whole or prepared fruit or vegetables (pineapple, melon, strawberries, grapes, tomatoes, oranges and blackberries.)	Foodlink
Napkins (SN)	Soiled paper napkins	GWDH
Post-consumer (POST)	Pieces of pizza crusts, French fries, mashed potatoes/gravy, ham scraps, home fries, chicken fingers, salad/dressing/grated cheese, pancakes.	GWDH
Kitchen Preparation waste (PREP)	Approximately 90% assorted melon rinds and seeds with balance consisting of rotting tomato, celery scraps, olives, kiwi peels, strawberry tops, carrot peelings and coffee grounds	GWDH
Salad mix (SM)	Rotting lettuce and bagged lettuce mixes	Foodlink
Yogurt and frozen desserts (YFD)	Greek yogurt (chocolate), Low-fat ice cream (blueberry), sorbet (mango), frozen greek yogurt (black cherry)	Foodlink

Food Sector blends		
Substrate	Contents	**Composition (% w/w)**
Food Service blend (SERVICE)	Post-consumer (POST) plate waste combined with kitchen preparation waste (PREP)	POST:PREP (80:20)
Café blend (CAFÉ)	Baked goods (BG) combined with coffee/filter paper (COF)	BG:COF(60:40)
Retail blend (RETAIL)	Combination of the fruit and veg waste (FVW), sweet dry goods (SDG), baked goods (BG), canned goods (CG) and yogurt and frozen desserts (YFD)	FVW:SDG: BG:CG: YFD (57:8:21:7:7)
Dairy manure co-digestion blend description		
Substrate	Contents	**Composition (% w/w)**
BG:M	Baked goods (BG) and dairy manure (M)	BG:M (30:70)
CAFE:M	Café mix (CAFÉ) and manure (M)	CAFÉ:M (22:78)*
CG:M	Canned goods(CG) and dairy manure (M)	CG:M (30:70)
FVW:M	Fruit and vegetable waste:manure (FVW:M)	FVW:M (30:70)
POST:M	Post-consumer :manure (POST:M)	POST:M (30:70)
PREP:M	Kitchen Prep waste (PREP) and dairy manure (M)	PREP:M (30:70)
RETAIL:M	Retail blend (RETAIL) and dairy manure (M)	RETAIL:M (30:70)
SDG:M	Sweet dry goods (SDG) and dairy manure (M)	SDG:M (30:70)
UDG:M	Unsweetened dry goods (UDG) and manure (M)	UDG:M (30:70)

*Sample preparation error resulted in a non-standard co-digestion ratio for this sample

4.2.2 Substrate characterization

Total solids (TS) dry matter and Volatile solids (VS) were determined according to the APHA Standard Methods 2540B and 2540E, which involves a gravimetric moisture determination at 105°C, followed by an ashing (ignition) of the dried sample at 550°C (APHA, 1998).

Crude protein was calculated from nitrogen (N) measurement using a heated block digestion with copper catalyst, followed by steam distillation into a boric acid solution per modified AOAC Method 984.13 (AOAC, 2012a). The sample was digested in sulfuric acid using copper sulfate as a catalyst. This converts bound nitrogen into ammonia, which was distilled and titrated with standard acid. A 6.25 conversion coefficient was used to calculate protein concentration from measured total Kjeldahl nitrogen (TKN).

Crude fat was measured via solvent extraction per modified AOAC 991.36 (AOAC, 2012b). Soluble fat-based materials are extracted from dried test samples via a two-step submersion treatment with hexane solvent. The crude fat content is determined by measuring weight after drying the hexane extracts.

Crude carbohydrates were calculated as the mass-balance difference of the crude fat, protein, moisture (*i.e.,* total solids) and ash determinations. This is a generalized approach for certain types of foods and biosolids. An example of this approach can be found in AOAC Method 986.25, where the general formula is presented as

Carbohydrate = total solids – (proteins + fat + ash) (AOAC, 2012c).

4.2.3 Biochemical methane potential assay

The biochemical methane potential (BMP) assay was first described by Owen et al. in 1979. With the rise in interest in anaerobic digestion it has been revised by others (Angelidaki et al., 2009; ASTM, 2008; Hansen et al., 2004) to improve repeatability and has become a standard method for measuring bio-degradability parameters of substrates. A total of 149 assays were prepared and conducted in 6 different phases. Microcrystalline, 20-µm, cellulose (SigmaCell type 20) was used as positive control samples across each phase. Inoculum was harvested from the post solid separated, effluent, from a full-scale, completely-mixed anaerobic digester operated at mesophilic temperatures that co-digested dairy manure with assorted food wastes (*i.e.,* whey, grease trap waste, and fruit and vegetable processing waste). Inoculum was pre-incubated at 37^{o}C for five days to minimize gas production from un-digested biomass. Samples were prepared to achieve an inoculum to substrate ratio (ISR) of 2 (gVS inoculum: gVS substrate added) to prevent biomass limiting kinetics (Jensen et al., 2011). Total solids content of all samples were less than 3% in prepared samples. Basic nutrient requirements for anaerobic microorganisms were provided by the dairy manure-based inoculum (Gustafson, 2000; Labatut et al., 2011). No additional external nutrients/ trace elements were added in order to evaluate the synergistic effects of co-digestion in providing these requirements. Measurements of pH for each sample prior to the start of the test ranged from 6.9 and 7.6. (Measurement at the end of the test ranged from 7.2 to 7.9.) Samples were flushed with N_2 to create an anaerobic environment and incubated at 37^{o} (± 1 C) with mixing at 10 seconds per minute. BMP vessels were 0.5L with working volumes ranging from 300-400mL. Bio-methane production was measured continuously using the AMPTS II (Bioprocess Control). The efficiency of the CO_2 fixing system was periodically verified by measuring CO_2 and CH_4 concentrations before and after entering the system using gas chromatography (TCD with helium carrier gas and HaysepQ packed column). Bio-methane production of substrates, blanks and controls were adjusted to standard temperature and pressure (STP) conditions (*i.e.,* 0°C, 1 atm). The BMP assay was conducted for 33 days, after which bio-methane production for all samples had reached a plateau. Blank samples containing only inoculum, were run in triplicate for each phase. Substrate bio-methane production was obtained by subtracting background methane production observed in the blanks.

Bio-methane potential, Bo

Substrate bio-methane production was normalized by VS to report observed bio-methane potential (B_o). In addition to B_o, to the standard specific methane yield reporting on a basis of VS added (mLCH_4/gVS), bio-methane potential was also reported based upon fresh mass of substrate digested (L_o) (m^3CH_4/t).

Theoretical methane yield, Bu and extent of degradation, fd

Theoretical methane potential *(B_u)* was calculated based upon the composition of each substrate, where proteins (based on $C_5H_7O_2N$) have a methane potential of 496 mLCH_4/g VS, carbohydrates (based on $C_6H_{10}O_5$) have a potential of 415 mLCH_4/g VS, and fat/lipids (based upon $C_{57}H_{104}O_6$) have a potential of 1014 mLCH_4/g VS (Buswell and Neave, 1930; Neilfa et al., 2015; Raposo et al., 2011). f_d can be calculated by the ratio of B_o to B_u, as follows (Raposo et al, 2011):

$$f_d = \frac{B_o}{B_u} \qquad \text{(Eq. 4-1)}$$

where f_d is the extent of degradation or substrate biodegradable fraction (decimal), and B_o and B_u correspond to the observed and theoretical bio-methane potential on a VS basis (ml CH_4/g VS added).

Hydrolysis rate coefficient

Hydrolysis is the rate-limiting step during the anaerobic digestion of particulate materials (Eastman and Ferguson, 1981). Thus for complex feedstock, parameters obtained from BMP tests should be directly applicable to characterize biodegradability in models such as the ADM1, *i.e.,* the extent of degradation, f_d and the apparent first order hydrolysis rate coefficient, k_h (Batstone et al., 2002).

The rate of hydrolysis of the biodegradable fraction of substrates was assumed to be first order and equivalent to the difference between the observed bio-methane potential, B_o, and the bio-methane production, B, at any given time, t.

$$\frac{dS}{dt} = -k_h(B_o - B) = -k_h(f_d B_u - B) \qquad \text{(Eq. 4-2)}$$

where S is the biodegradable substrate and t is time (d). The extent of degradation (f_d) and apparent hydrolysis rate coefficient *(k_h)* where estimated using the secant method of Aquasim 2.1g that simultaneously fits these two parameters from the BMP data (Gustafson, 2000; Reichert, 2014).

Co-digestion performance index (CPI) and co-digestion rate index (CRI):

AcoD can result in increased bio-methane production when the organic load of the combined substrate is higher than that of the original substrate. However, the combination of substrates can also result in synergistic effects. Synergistic effects may arise from dilution of inhibitory intermediaries, addition of valuable nutrients that result in increased bio-degradability, and/or a change in the microbiome that results in an enhanced metabolism. Labatut et al. (2011) suggested comparing the bio-methane potential of a co-digested substrate with the weighted sum of the single substrate bio-methane potentials as a measure of synergistic or antagonistic interactions. A co-digestion performance index *(CPI)* was calculated as the ratio of the bio-methane potential of the co-digestion blend ($B_{o_{i,n}}$) to the weighted average ($\overline{B_o}_{i,n}$) based upon VS content ($\%VS$) of the individual substrate bio-methane potentials ($B_{o,i}$):

$$CPI_{i,n} = \frac{B_{i,n}}{\overline{B_o}_{i,n}} = \frac{B_{i,n}}{\sum_i^n \%VS_i Bo_{,i}} \qquad \text{(Eq. 4-3)}$$

where substrates *i* through *n* are co-digested such that $\sum_i^n \%VS_i = 1$. Thus, a CPI > 1 indicates a synergistic effect of co-digestion and CPI < 1 indicates an antagonistic affect.

Similarly, a co-digestion rate index *(CRI)* was calculated to compare the apparent hydrolysis rate coefficient for co-digested substrates, $k_{h_{i,n}}$with the rate obtained from a predictive curve obtained by adding the methane production curves of the individual substrates. The rate coefficient for the sum of two cumulative exponential decay curves could not be determined mathematically. Therefore a simulation was used to determine an appropriate relationship between the rate coefficients of individual curves and that of the curve resulting from summing them; see Supplementary Material S.2 for details.1000 co-digestion blends were simulated with parameters (k_h and B_o) within the range of the data. The best estimate of the combined hydrolysis rate coefficient was obtained by the geometric mean $(\overline{Gk}_{h_{i,n}})$ of the individual substrate hydrolysis rate coefficients. Thus, the co-digestion rate index was calculated as the measured rate coefficient $(k_{h_{i,n}})$ over the predicted rate coefficient $\overline{(Gk_{h_{i,n}}})$:

$$CRI_{i,n} == \frac{k_{h_{i,n}}}{\overline{Gk}_{h_{i,n}}} = \frac{k_{h_{i,n}}}{\sum_i^n \exp((\%VS_i * B_{o,i}) * \ln(k_{h,i})) / \sum_i^n (\%VS_i * B_{o,i})} \quad \text{(Eq. 4-4)}$$

where substrates *i* through *n* are co-digested such that $\sum_i^n \%VS_i = 1$. The maximum bio-methane production for each constituent is the bio-methane potential of the substrate $(B_{o,i})$ weighted by the *%VS* of the substrate in the blend. A CRI >1 indicates that co-digestion had an accelerating effect on apparent hydrolysis rate and a CRI<1 indicates that co-digestion had a slowing effect.

4.3. *Results and Discussion*

4.3.1 Substrate characterization

Characterization of the waste categories is shown in Table 4-2. Although all samples would be disposed of as solid wastes, several samples (canned goods (CG), spent coffee grounds (COF), fruit and vegetable waste (FVW), salad mix (SM) and kitchen prep waste (PREP)) had solid content <30%. All food wastes showed VS/TS ratios over 90 % (vs. 83.6% for manure). Measured carbohydrate content ranged from 61% to 85% of TS. Protein constituted 10% to 20% of TS for most samples (with SM showing a higher content and PREP waste a lower content). Post-consumer waste (POST) and stale baked goods (BG) contained the highest lipid content.

Table 4-2: Substrate characterization

Substrates	%TS/ FM	%VS/ TS	TVS (%VS/ FM)	Composition of solids (TS)[a]			
				% ash/ TS	% crude lipids (CL)/ TS	% crude protein (CP)/ TS	% carbohydrate / TS
Baked goods (BG)	91.6%	97.9%	88.9%	3%	11%	10%	76%
Canned goods (CG)	10.5%	90.7%	9.6%	9%	2%	15%	74%
Coffee grounds (COF)	29.3%	99.3%	29.1%	1%	4%	17%	79%
Fruit and vegetable waste (FVW)	7.7%	93.3%	7.1%	7%	0%	10%	83%
Post consumer (POST)	46.6%	97.1%	45.2%	3%	21%	17%	59%
Preparation waste (PREP)	14.3%	100.0%	14.3%	0%	3%	15%	82%
Salad mix (SM)	3.8%	90.6%	3.4%	11%	2%	23%	65%
Soiled napkins (SN)	91.1%	100.0%	91.1%	NA	NA	NA	NA
Sweet dry goods (SDG)	92.7%	95.0%	88.0%	5%	2%	11%	82%
Unsweetened dry goods (UDG)	92.4%	97.8%	90.4%	2%	1%	12%	85%
Yogurt and frozen deserts (YFD)	30.9%	97.9%	30.3%	2%	5%	14%	79%
Manure (M)	10.2%	83.6%	8.5%	16%	1%	14%	69%

[a]Rounding error may lead to nutrients not summing to 100% total solids
NA=not measured
All samples were measured in triplicate.

4.3.2 BMP test results:

Key bio-methane kinetic parameters are summarized in Appendix C Table C-1. Bio-methane potential of the cellulose controls across all phases showed good agreement with expected results measuring 353 (σ= 44) $mLCH_4/gVS$ (n=15) and f_d of 85%. The average apparent first-order hydrolysis rate coefficient for cellulose of 0.32 d^{-1} showed good agreement with Jensen et al. who reported a k_h based upon methane production of 0.36 d^{-1} at an ISR of 2.

<u>Bio-methane potential</u>

Dairy manure resulted in a B_o of 238±19 $mLCH_4/gVS$ (n=12), which compares remarkably well with previously reported results (El-Mashad et al., 2010; Hoffman et al., 2008; Labatut et al.,2011; IPCC,1997)(Fig. 4-1) Food service waste (SERVICE) resulted in 496 $mLCH_4/gVS$ which was the highest B_o observed; manure co-digested with kitchen prep waste (PREP:M) resulted in the lowest B_o (165 $mLCH_4/gVS$ (σ=19)) (Fig. 5-1). All food waste substrates showed higher B_o than dairy manure when digested alone. Substrates with high lipid content, such as POST and BG, resulted in higher B_o. Raw fruits and vegetables (FVW) resulted in higher average B_o than processed fruits and vegetables (CG) (although this was attributed to the substrate composition as both substrates were nearly completely bio-degraded (Table 4-3)). Both fruit and vegetable substrates produced more methane than the purely vegetable, salad mix (SM) substrate. However, only SM and CG showed a statistical difference based upon a pairwise student t-test at $p<0.05$. This was attributed in part to the large variability observed in the FVW results. Several other substrates presented high variability notably, FVW:M, UDG and POST waste which reported relative standard deviations (RSD) of 30%, 27% and 19% respectively (where ($RSD = \sigma/\mu$)). Potential sources of variability include substrate non-homogeneity, clumping of pulverized samples (UDG) and process inhibition or nutrient deficiencies related to substrate heterogeneity.

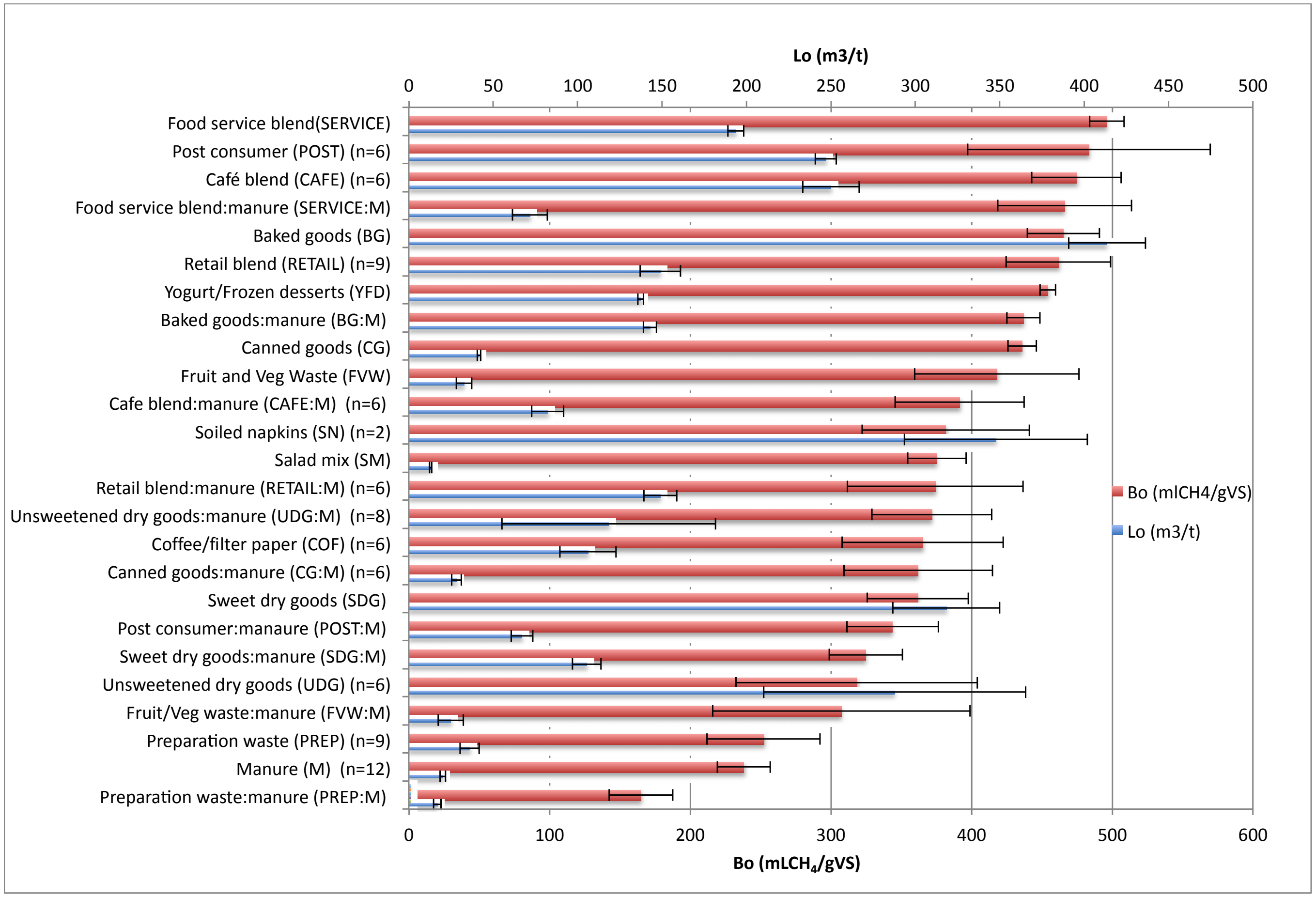


Figure 4-1: Standard bio-methane yield (Bo) for the substrates tested (mL CH4/g VS) shown in red with axis below graph. Methane yield per unit mass (Lo) (m3 CH4/tFW) shown in blue with axis above graph. Substrates were tested in triplicate (n=3) unless otherwise noted. Error bars represent one standard deviation (σ).

Both of the dried goods samples performed similarly (pairwise student t-test $p>0.05$). SDG, demonstrated slightly higher degradability than UDG which was unexpected as it contained higher concentrations of glucose, fructose and crude lipids (Appendix C Table C-1). Both dried goods substrates demonstrated lower bio-methane potentials than the fruit and vegetable substrates, although statistical difference was only shown with the dried goods samples and CG ($p < 0.05$).

Results were shown to be reasonable when compared to similar substrates found in the literature. Gunaseelan (2004) tested 24 fruit and vegetable wastes collected in South India and found substantial differences among the varieties of FVW and even among different parts of the plant with methane yields ranging from180-732 $mLCH_4/gVS$. Cabbai et al. (2013) analyzed samples collected from Italian canteens, supermarkets, restaurants, fruit/vegetable markets and bakery shops. Their supermarket and market waste contained only fruits and vegetables and ranged from 99 to 363$mLCH_4/gVS$. This was lower than the results for FVW in the current study, however, the composition of the wastes differed. The results reported by Cabbai et al. (2013) for bakery waste showed good agreement with the BG sample although the pastries and fillings comprising the Italian bakery waste reported a higher lipid content. The bio-methane potential of the Italian food service wastes were higher (571 to 675$mLCH_4/gVS$) than the SERVICE (496 $mLCH_4/gVS$) and POST (483$mLCH_4/gVS$) samples in this study which is again attributed to temporal and regional variation. Menardo and Balsari (2012) tested several waste substrates from the European retail market. This included a dairy waste substrate consisting of waste milk, yogurt and cheese, which reported a bio-methane production higher (545 $mLCH_4/gVS$) than YFD in this study (454$mLCH_4/gVS$) which was attributed to the lower fat content of the U.S. dairy products. Menardo and Balsari's results for stale bread were consistent with the UDG sample in this study. There are limited reports of anaerobic digestion of coffee production waste and variation in substrate characteristics (*i.e.,* TS, % lipid) can be observed in these studies (Dinsdale et al., 1996; Qiao et al., 2013). However, Neves et al (2006) tested several blends containing coffee and coffee substitutes and reported bio-methane production consistent with the coffee ground/filter paper sample (COF) in this study.

Results for bio-methane production were also expressed on a fresh weight (FW) basis (Fig. 4-1). This illustrates the large affect that moisture content can have on substrate methane potential per unit mass. While the %VS/TS ranged from 90% to 100% for the commercial food waste substrates, the large variation in solids content resulted in TVS ranging from 3.4% to 90.1% of FW. This had a large effect on bio-methane yield per unit mass (L_o). Substrates with high solids content (baked goods, soiled napkins and dry goods) result in L_o that were an order of magnitude higher than those of substrates with higher moisture content.

<u>Theoretical methane yield (Bu) and extent of biodegradation (fd)</u>

The extent of bio-degradation was calculated via Eq. 1 and compares the observed bio-methane potential (B_o) to the theoretical bio-methane potential (B_u) (Fig. 4-2). Several substrates showed an extent of bio-degradation (f_d) greater than 95%. These substrates were observed to be rich in readily hydrolysable carbohydrates (decayed FVW and processed CG) and fats (YFD, BG, café and RETAIL). The lowest conversion efficiencies was manure (54%), which was attributed to a higher content of lignin or other recalcitrant carbon than food wastes. Kitchen preparation waste (PREP) also resulted in low bio-degradability (56%); this could be due to the lignin content in the seeds and rinds of the preparation waste, nutrient deficiencies or inhibitory compounds.

Buswell's equation is based upon a balanced redox equation where the substrate (and water) is completely converted to CH_4 and CO_2, therefore B_u should always be greater than the observed B_o due to cellular synthesis and incomplete digestion. Raposo et al. (2011) estimated the organic matter consumed in microbial biomass to be near 15% for reference carbohydrate and proteinaceous substrates, but cite literature ranging from 3%-15%. In this study, some degradation extents for individual assays were observed near or greater than 95% (CG, RETAIL, FVW, POST). This was attributed to heterogeneity in the sample, which may have resulted in a difference between the sample characterization (and in turn the calculated theoretical yield) and the tested substrate. This is supported by the large variability observed in the individual sample results. Error in determining lipid, protein and carbohydrate content of the substrate as well as the formulae for model nutrient compounds could also be a factor.

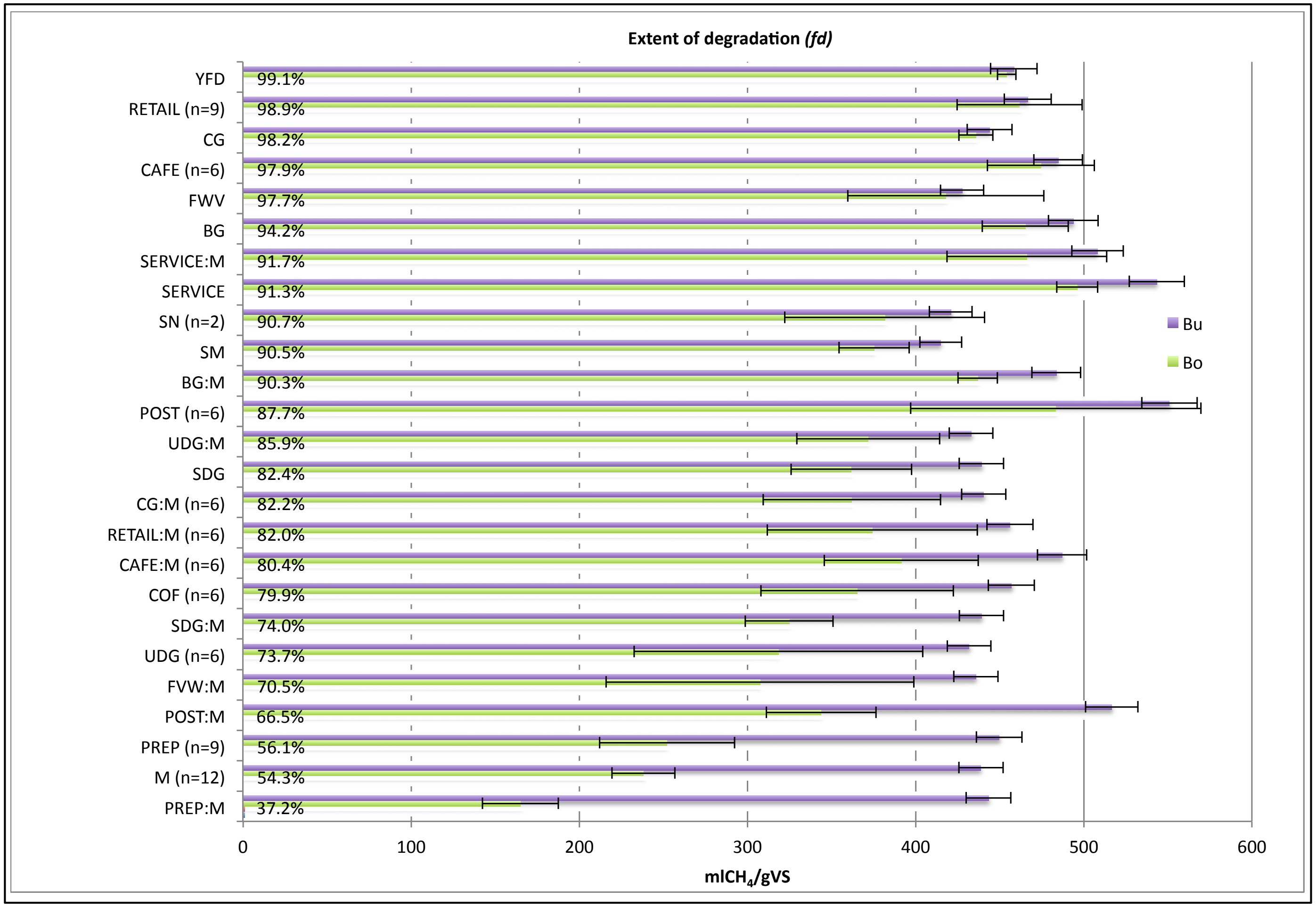


Figure 4-2: Comparison of observed bio-methane potential (B_o) to theoretical bio-methane yield (B_u). Error bars indicate standard deviation of the experimental data and estimated error of the theoretical calculation of 3% based upon method error estimation. (The ratio of B_o/B_u is the extent of degradation (f_d) and is shown as a percentage in Table C-1 in Appendix C.

Apparent hydrolysis rate coefficient (k_h)

Apparent first-order hydrolysis rate coefficients ranged from k_h= 0.14 (0.01) d^{-1} for coffee and filter paper (COF) to k_h=0.64 (0.05) d^{-1} for salad mix (SM) (Table 4-3). Several substrates (CEL, SM and UDG) showed a high standard error indicating a poor fit to the first-order decay model used for parameter estimation.

Table 4-3: Apparent hydrolysis rate coefficients and standard errors.

Substrate (w:w:w)	k_h	**s.e.***
Manure (M) (n=12)	**0.19**	**0.011**
Cellulose (n=12)	**0.32**	**0.032**
Coffee/filter paper (COF) (n=6)	0.14	0.009
Sweet dry goods (SDG)	0.20	0.003
Baked goods (BG)	0.26	0.007
Post-consumer (POST) (n=6)	0.27	0.016
Canned goods (CG)	0.32	-
Fruit and Veg Waste (FVW)	0.34	0.01
Yogurt/Frozen desserts (YFD)	0.40	0.011
Unsweetened dry goods (UDG) (n=6)	0.47	0.033
Preparation waste (PREP) (n=9)	0.48	0.027
Salad mix (SM)	0.64	0.049

*s.e. is the standard error in estimating the apparent hydrolysis rate coefficients

Generalizable conclusions regarding the impact of substrate characteristics on hydrolysis rate are difficult to draw. FVW and CG showed similar rate coefficients of 0.32d-1 and 0.34 d-1 respectively. However the salad mix (SM) and the kitchen preparation waste (PREP) showed significantly faster degradation profiles. While SM had a higher protein concentration suggesting an improved C:N ratio, PREP had a lower protein concentration, yet both showed higher hydrolysis coefficients than FVW and CF. Interestingly, despite similar compositions, unsweetened dried goods (UDG) resulted in a higher apparent hydrolysis coefficient than sweetened dry goods (SDG), suggesting that other factors beside composition, play a role in digestion kinetics.

Co-digestion parameters

Co-digestion performance index (CPI)

Co-digestion performance indexes (CPI) ranged from 0.68 for PREP:M to 1.21 for UDG:M (Fig. 4-3) Nine of the 13 co-digested samples indicated a synergistic affect, based upon mean B_o values, while 4 indicated an antagonistic affect. However, all but three of the samples did not show an effect that was statistically different from the weighted average of the individual substrates (CPI=1). Food service blend:manure (SERVICE:M) and canned goods:manure (CG:M) showed a statistically significant synergistic effect. This is presumed to be due to synergistic mechanisms such as the buffering of volatile solids in AcoD between manures and C-rich wastes as described (Mata-Alvarez et al.) 2014. The reason for the highly antagonistic effect observed for PREP:M was not evident. Near neutral pH at the end of the assay did not indicate a build-up of VFA and the high apparent hydrolysis rate coefficient observed for PREP appeared to be moderated by the addition of manure resulting in a reduction in k_h for the PREP:M mixture. Toxic or inhibitory compounds in the PREP waste are suspected although a review of the literature did not reveal any insight; thus further characterization and testing is suggested. Nutrient deficiency is also a potential cause to consider.

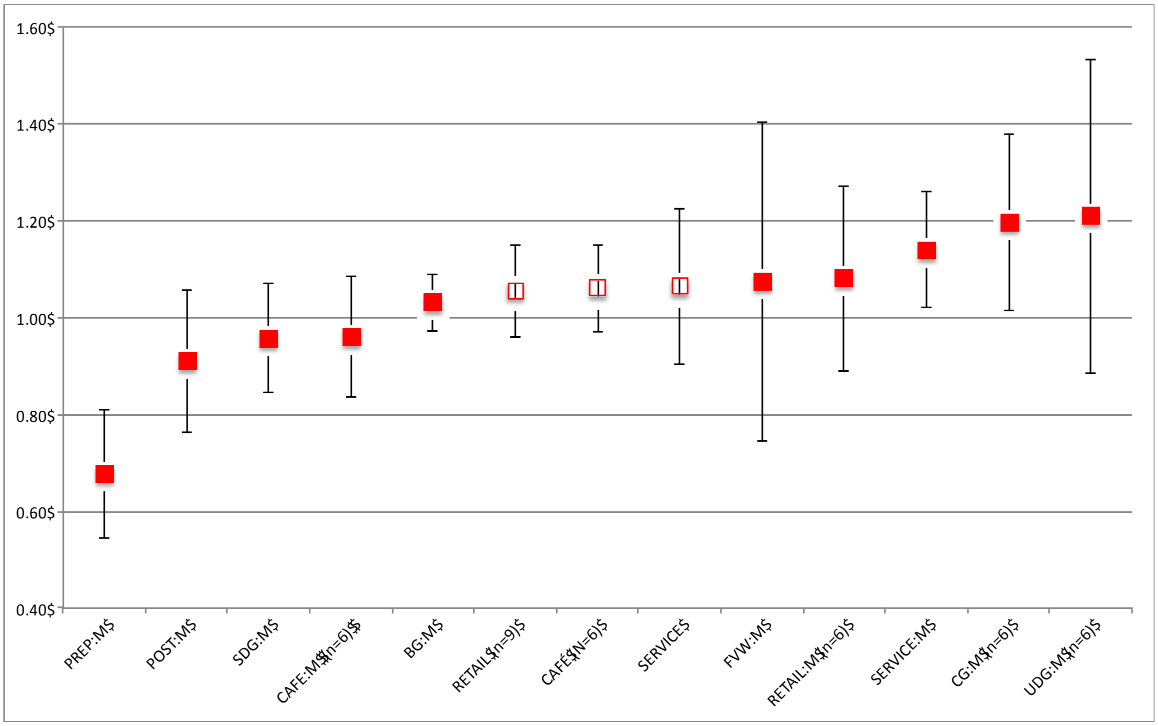


Figure 4-3: Co-digestion performance index (CPI) of co-digestion substrates. CPI>1 indicates synergistic effect, CP<1 indicates antagonistic effect. Indicates co-digestion with manure indicates food waste co-digestion blends. Error bars indicate standard deviation.

Co-digestion rate index (CRI)

The range of apparent hydrolysis rates for co-digested substrates ranged from 0.19 d^{-1} for FVW:M to 0.44 d^{-1} for RETAIL:M. Apparent hydrolysis rate coefficient for the co-digested substrates was higher than the geometric weighted average of the individual substrate coefficients for 10 of the 12 co-digested substrates (Fig. 4-5). Only FVW:M and UDG:M resulted in co-digestion rate indices below 1 (0.80 and 0.95 respectively). The co-digestion indices were for RETAIL (1.68) and CAFE blends (1.59) were the highest.

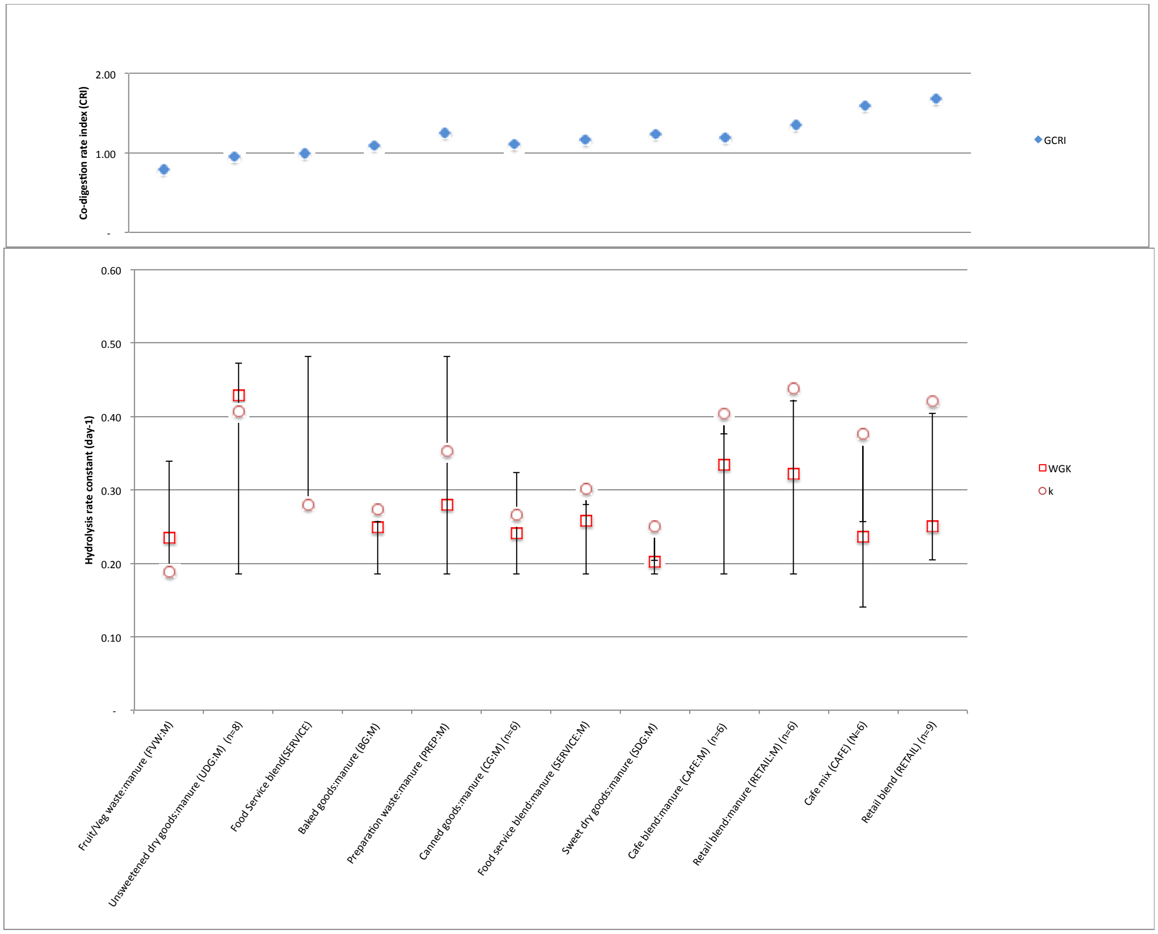


Figure 4-4: Co-digestion rate index (CRI) of co-digestion substrates (upper portion of figure). Hydrolysis rate coefficients of co-digestion substrates (k_h) compared to weighted average of individual substrate hydrolysis rates ($\overline{(k_h)}$) (lower portion of figure). Indicates substrates co-digestion with manure; indicates food waste co-digestion blends. (Standard error associated with estimating hydrolysis rate coefficients could not be used to estimate statistical significance.)

These results are in agreement with the observations of Astals et al. (2014) who reported a general improvement in process kinetics without a significant change in biodegradability when comparing varying co-digestion mixtures of pure and slaughterhouse carbohydrates, protein and lipids. They attributed their results to mitigation of inhibitory compounds, particularly dilution of fat concentration and mitigation of long-chain fatty acids (LCFA) inhibition. The high *CRI*'s observed in this study for the RETAIL (1.68) and RETAIL:M blends (1.16) may be attributed to this effect as lipid rich baked goods (BG) were a constituent of both RETAIL and RETAIL:M blends. However, other high lipid content substrates did not exhibit such a significant kinetic synergism (*i.e.,* BG:M and SEVICE (POST:PREP)). It is worth noting that BG and POST, although high in lipid content for commercial food wastes (11% TS and 19% TS respectively) have significantly lower content than the pure lipids or olive oil used in the Astals et al. (2014) study thereby resulting in less LCFA-related inhibition to mitigate. Another possible cause for the strong synergisms observed in RETAIL and RETAIL:M may be the supply of nutrients or trace elements from the co-substrates. Whereas, addition of a nutrient medium as cited in the BMP protocol referenced by Astals et al. may have masked this type of synergy. As a further example, combining BG with COF, both of which had higher lipid content resulted in a higher apparent hydrolysis rate coefficient (in the CAFÉ blend) than either of the individual substrates ($k_{h,\mathrm{BG,COF}}$=0.38 vs. $k_{h,\mathrm{BG}}$=0.26 and $k_{h,\mathrm{COF}}$=0.14) and a co-digestion ratio index of 1.59. The significant synergism observed may be due to dilution of another inhibitory compounds such as the unidentified inhibition observed in digesting coffee grounds by Lane (1983). Thus, the use of actual food waste substrates, along with information on their micro- and macro-nutrients is important to uncovering possible causes of synergism (or antagonism) observed in co-digestion mixtures.

4.4. *Conclusions*

Bio-methane potential was a result of substrate nutrient composition as well as biodegradability. Substrates with high fat content resulted in higher bio-methane production. Substrates rich in readily hydrolysable carbohydrates and fats showed high bio-degradability. Co-digestion resulted in bio-methane production close to that of the weighted average of the individual substrates ranging from-5%/+20% on average. Co-digestion apparent hydrolysis rates showed an increase in 10 of 12 substrates which was attributed to dilution of inhibitory effects and improved nutrient balances as substrate complexity increased. Macro-nutrient composition alone was not sufficient to explain synergistic impacts pointing to other factors such as provision of micro-nutrients, build up/dilution of inhibitory compounds.

Chapter 5 : Climate change impacts of food supply chain resources based upon feedstock characteristics: Application of the ORCAS model to New York State

5.1. *Introduction*

In Chapter 2, analyses of climate change impacts were presented for two emerging FSC utilization pathways. While the facilities studied in many ways were representative of typical NYS implementation of these technologies, it was noted that net environmental impacts were highly dependent upon the feedstock processed and the alternative treatment scenario for that feedstock. Thus making it difficult to extend that work to analyze the climate change impacts associated with the variety of FSC resources generated in the state (as analyzed in Chapter 1 and characterized in Chapter 3). Therefore the objective of this chapter is to build upon the previous chapters by providing a tool to assess the climate change impacts of various FSC resources relative to alternative treatment pathways for those resources.

Several studies have compared the GHG impacts, of alternative treatments for municipal solid waste (MSW) and among significant factors cited is the influence of MSW composition (Christensen et al., 2009a; Gentil et al., 2010). Some studies have specifically considered the food waste constituent of municipal solid waste (MSFW) or the organic fraction of municipal solid waste (OFMSW) (Baky and Eriksson, 2003; Boldrin et al., 2011; Kim and Kim, 2010; Laurent et al.; 2014; Levis and Barlaz, 2011). However, resources leave the food supply chain (FSC) as wastes at every stage and these resources have unique characteristics that may influences treatment options.

A US industry group study estimated that commercial and industrial food waste constituted 66% of disposed food waste (BSR, 2012). FSC resources generated at the industrial level (i.e., food processing plants) include by-products or rejects from food manufacturing processes. In the commercial sector, retail establishments may generate source-separated waste streams from different operations within a store or at different types of stores. Food preparation and service at restaurants, institutions or businesses are also a source of commercial food waste.

Because these sectors generate large quantities of food waste, they are often targeted by waste management policies such as landfill disposal bans (ARNI, 2014). To choose among available options for waste utilization, data on the climate change impacts for FSC resources are important to inform a balance of environmental, social and economic considerations. This study provides an open source model for greenhouse gas (GHG) impacts based upon resource characteristics. To demonstrate the model, point estimates were calculated for several FSC resources, results were compared to those for the generic category of MSFW. Generalized expressions were presented to relate utilization emission factors to key FSC resource characteristics. Finally, the main sources of uncertainty were analyzed to assess the sensitivity of model results which will inform commercial and industrial food waste generators, policy makers and developers on the environmental impacts of managing specific FSC resources. The study also provides insight to lifecycle practitioners who apply waste utilization emission factors based on the impact that specific resources or resource characteristics have. The models provided can be used to calculated FSC resource specific impacts to be incorporated into multi-criteria analysis of waste management alternatives. This work advances climate change modeling of food waste climate impacts by incorporating many of the recent findings in this evolving field.

The general motivating research question is:

How do FSC resource characteristics affect the GHG impacts associated with available utilization options?

5.2. *Methods:*

5.2.1 Model development

The ORCAS (Organic Resource Climate Assessment Simulator) model was created in the programming language R and consists of several sub modules. Input files provide key characteristics for a set of FSC resources into the model and utilization pathway modules estimate GHG impacts for each waste utilization pathway. Additional scripts and files provide outputs, higher-level functions and utilities, which are described in the Readme.md file in Github

(https://github.com/graySquirrel/foodwasteTreatmentSim/blob/master/README.md)

5.2.2 Food Supply Chain (FSC) Resources

Several types of resources generated at various stages of the food supply chain were identified as inputs to the model (Table 5-1). These materials were earlier identified in Chapter 2 as being available in significant quantities in New York State, and thus are relevant to analysis of viable conversion options, including landfills, composting, anaerobic digestion and animal feed.

Table 5-1: Descriptions of Food Supply Chain (FSC) resources

FSC resources	Description	Sector
MSW FW	The food waste constituent of municipal solid waste (reference).	MSW
Apple pomace	A by-product of apple juice extraction.	Industrial
Brewer's spent grains (wet)	The by-product of the beer-brewing industry consisting mostly of barely, but some corn and/or rice may be included depending on the source of the grains.	Industrial
Grape pomace	The solid remains after pressing (sometimes called marc), may contain skins, pulp, seeds, and stems of the grape.	Industrial
Tomato pomace	A by-product of tomato food processing such as juice, ketchup, sauces or soups.	Industrial
Whey	A by-product of yogurt or cheese making.	Industrial
Baked goods	Based upon samples containing stale bagels, muffins and donuts.	Retail
Canned goods	Damaged cans removed from the shelf (crushed tomatoes, diced tomatoes, green beans, beets, chicken broccoli soup, cream of chicken soup, cheese pot pie soup, baked beans, papaya, pineapple chunks, tuna fish and mandarin oranges).	Retail
Coffee grounds and filter paper	Spent coffee grounds (medium roast) and coffee filter paper	Retail
Dry goods	Assorted grains removed from the shelf (rice, oatmeal, bread crumbs, cream of wheat).	Retail
Salad	Rotting lettuce and bagged lettuce mixes.	Retail
Sweet cereals	Assorted breakfast cereals removed from the shelf (Cocoa O's©, Cap'n Crunch©, Shredded Wheat©, Lucky Charms©, Chex©, Frosted Flakes©,quick oats, pasta, Cliff© cereal bar).	Retail
Mixed produce	Approximately 50% rotting bagged lettuce and 50% rotting whole or prepared fruit or vegetables (pineapple, melon, strawberries, grapes, tomatoes, oranges and blackberries).	Retail
Refrigerated and frozen dairy	Assorted yogurts and frozen desserts (Greek yogurt (chocolate), Low-fat ice cream (blueberry), sorbet (mango), frozen greek yogurt (black cherry)).	Retail
Post-consumer	Cafeteria plate waste (pieces of pizza crusts, French fries, mashed potatoes/gravy, ham scraps, home fries, chicken fingers, salad/dressing/grated cheese and pancakes.	Food service, retail
Preparation waste	Kitchen preparation waste consisting of approximately 90% assorted melon rinds and seeds with balance consisting of rotting tomato, celery scraps, olives, kiwi peels, strawberry tops, carrot peelings and coffee grounds.	Food service, retail

Characterization data for the commercial FSC resources (comprising retail and food service) were taken from Ebner et al. (2016) and supplemented with data from literature. Industrial FSCR characterizations were gleaned from the literature (Table 5-2). FSC resource characteristics are distinct from resources generated at other stages, however the categories are broad and subject to heterogeneity and variability. Thus the FSC resource characteristics should be viewed as point estimates constituting characteristics representative of a given FSC resource but not to imply a level of statistical certainty.

Table 5-2: Published characteristics of FSC resources and associated references.

FSCR	Total Solids (TS) (%FW)	Volatile Solids (VS) (%TS)	Bio-methane potential (B_o) (ml/g VS)	Total Kjeldahl (N) TKN(mg/kg)	Crude Lipids (CL) (% TS)	Crude Protein (CP) (%TS)	Cabo-hydrate (%TS)¥	Potassium (K) (mg/kg)	Phosphorus (P) (mg/kg)
MSWFW	30%[1]	90%[1]	334[1]	8900[2]	14%[3]	13%[3]	67%[3]	3300[2]	1900[2]
Apple Pomace	26%[4]	90%[4]	228[5]	6800[4]	4%[4]	6%[4]	75%	2597[4]	303[4]
Brewers spent grains	21%[6]	91%[6]	446[6]	8000[6]	11%[7]	23%[7]	62%[7]	200[8]	1300[8]
Tomato Pomace	32%[9]	98%[9]	218[9]	1069[9]	6%[10]	19%[10]	71%[10]	109[11]	117[12]
Whey	6%[13]	91%[13]	240[14]	650[15]	1%[13]	10%[13]	72%[13]	150[13]	80[13]
Baked goods[16,17]	92%	98%	465	14656	11%	10%	76%	1400	1200
Canned goods[16,17]	11%	91%	436	2520	2%	15%	74%	1380	370
Coffee grounds and filter paper[16,17]	29%	99%	365	7970	4%	17%	79%	1435	2670
Dry goods[16,17]	92%	98%	318	17741	1%	12%	85%	2010	1630
Mixed produce[16,17]	8%	93%	418	1232	0%	10%	83%	2315	330
Refrigerated and frozen goods[16,17]	31%	98%	454	6922	5%	14%	79%	1500	1100
Salad [16,17]	4%	91%	375	1398	2%	23%	65%	2300	300
Sweet Cereals[16,17]	93%	95%	362	16315	2%	11%	82%	1170	740
Post consumer[16,17]	47%	97%	483	13421	21%	17%	59%	3200	1200
Prep waste[16,17]	14%	100%	252	458	3%	15%	82%	1200	977

[1]Levis and Barlaz, 2014, [2] Banks et al., 2011; [3]Estimated from data provided in Levis and Barlaz, 2014; [4]Dhillon et al., 2012; [5]Frear et al., 2005 (estimated from peels from Lane, 1984); [6]BDI, 2015; [7]Mussato et al., 2005; [8]Alaska cooperative extension, 2015; [9]Dinuccio et al., 2010; [10]DelValle et al., 2006; [11]Elbadrawy and Sello, 2011; [12]Abdollahzadeh et al., 2010; [13]deWit et al., 2001; [14] Ghaly et al.,1997; [15]Rico et al., 2015; [16]Ebner et al., 2016 (TS,VS and B_o); [17] Based upon estimates values from USDA, 2015

NA not available

¥Calculated as the remainder where % carbo=1-%ash-%fat-%protein

5.2.3 Lifecycle framework

A lifecycle assessment (LCA) methodology was applied with the goal of providing a comparative assessment of climate change impacts associated with various pathways for FSC resource utilization. The functional unit (FU) chosen was the treatment of 1 ton of FSC resources. Although co-processing of FSC resources with other substrates is common to optimize process performance (based on C/N ratio, moisture content etc.) or economics (ie. tipping fees), this selection of FU is consistent with other studies where the goal was to understand the impact of a particular process input (e.g., Baky & Eriksson (2003)).

Many variations of technologies exist for treatment of FSC resources. Four representative treatment pathways that are commercially available for FSCRs in New York State were selected for modeling. They include: landfill (LF), anaerobic digestion (AD), composting (C) and direct diversion to feed animals (AF).

The system boundary was taken to be the gate of the treatment facility covering all exchanges with the ecosphere (and the technosphere for co-products) until a period of 100 years after disposal. After careful consideration, transportation of the FSC resources to the facility was not included. Primarily it was reasoned that transportation was not considered to be a factor of treatment pathway (as much as geography) and thus outside the scope of this analysis. Additionally, as shown in Chapter 2 and other studies (Bernstadt et al., 2012) transportation of FSC resources did not have a significant climate change impact. Furthermore, since the FSCR's were assumed to be source-separated, screening or sorting operations were not included, nor were any losses due to those functions assumed. The system was expanded to include the net impact of displacing goods or services that result from treatment of the FU.

Climate change impacts were evaluated in terms of carbon dioxide equivalents (CO_2e) using the 100-year Global Warming Potential (GWP) factors without climate feedback of 28, 265 and 1 for CH_4, N_2O and fossil CO_2, respectively (IPCC, 2013). Biogenic CO_2 released during composting or combustion of biogas were considered neutral with respect to GWP (Christensen et al., 2009). Biogenic carbon that is sequestered for longer than the 100-year time frame for global warming was counted as a negative flux.

Lifecycle modeling of waste management has evolved over time and with varying regional focuses (Gentil et al., 2010). Every attempt has been made to incorporate recent developments in waste management modeling and when possible factors relevant to U.S. waste treatment pathways were used. Consistent with the comparative objective of this LCA, particular care was taken to provide consistent system boundaries and life cycle accounting across the treatment pathways.

5.2.4 Treatment pathways

Individual treatment pathways functions were created and are discussed below. Emission factors common to several treatment pathways and their sources are shown in Table 5-3.

Table 5-3: Common emission factors

Global Factors	Units	Value	Low	High	Source
Provision of diesel fuel	$kgCO_2e/L$ diesel	0.45	0.4	0.5	Fruergaard et al. 2009
Diesel fuel combustion	$kgCO_2e/L$ diesel	2.72			ANL, 2015
Grid non-baseload emissions	$kgCO_2e/MWh$	692.15	918.79	537.18	U.S. EPA, 2014
Production N fertilizer	$kgCO_2e/kgN$	8.85	4.7	13	Boldrin et al., 2009
Displaced application of N fertilizer	$kgCO_2e/kgN$	5.40			Ebner et al., 2015
Diesel fuel for grinding[a]	L/t	2.65	2.5	3.3	Bernstadt and Jansen, 2012
Production P fertilizer	$kgCO_2e/t$	1.80	0.52	3.09	Boldrin et al., 2009
Production K fertilizer	$kgCO_2e/t$	0.96	0.38	1.53	Boldrin et al., 2009
Production of peat	$kgCO_2e/kg$ peat	970	388	1197	Boldrin et al., 2009
Indirect emission factor	kg N_2O-N/kg N	0.01	0.002	0.05	IPCC, 2006

[a]Inventory factor used in AD and Animal Feed treatment pathways

5.2.4.1 Landfill treatment pathway

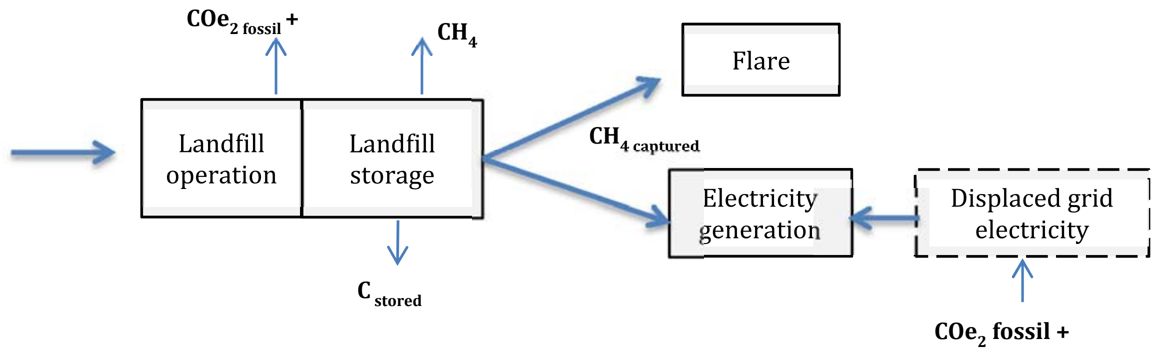


Figure 5-1: Landfill pathway system diagram.

Landfill emissions consist of those due to operation of the landfill, methane released to the atmosphere and carbon stored long-term in the landfill and the net impact of methane captured by a landfill gas (LFG) recovery system including displacement of grid electricity (Fig. 5-1).

Methane generation in the landfill was estimated based upon a first order decay model as presented in the U.S. EPA LandGEM v.3.02 model evaluated over 100 years (U.S. EPA, 2015a)

$$Q_{CH4} = \sum_{1=1}^{10} \sum_{j=0.1}^{1} kL_o \left(\frac{M_i}{10}\right) e^{-kt_{ij}} \qquad \text{(Eq. 5-1)}$$

where:

- Q_{CH4} = annual methane generation in the year of the calculation (m3/year)
- i = 1 year time increment
- n = (year of the calculation) - (initial year of waste acceptance)
- j = 0.1 year time increment
- k = methane generation rate (year^{-1})
- L_o = potential methane generation capacity (m^3/t)
- M_i = mass of waste accepted in the i_{th} year (t)
- t_{ij} = age of the j_{th} section of waste mass M_i accepted in the i_{th} year

The LandGEM model was modified to estimate specific FSC resource methane generation as follows:

- k = methane generation rate for the category of food waste provided by De la Cruz and Barlaz (2010).
- Although Ebner et al. (2016) observed variation in methane generation rates across FSC resources in lab bio-methane potential (BMP) assays, a correlation between lab rate constants and landfill rate constants was not available. Since k also encapsulates landfill conditions such as the availability of nutrients, pH, temperature and moisture, which may account for more variation than resource characteristics, the range of values provided by Levis and Barlaz (2011) for MSWFW was used to investigate sensitivity of results to all of the factors influencing k.
- M = 1t FSC resource placed in the landfill at t=0 with no additional waste added until t=100 years.
- L_o= calculated from the bio-methane potential (ml CH_4/g VS) of the FSCR and a correction factor (*Cf*) per the experiments of Cho et al. (2012). (Note: this can also be viewed as uncertainty in B_o for a given FSCR.)

where

$$L_o = Cf * B_o * VS \qquad \text{(Eq. 5-2)}$$

Microbial oxidation of methane as it passes through aerated parts of the landfill cover soil was accounted for by an oxidation factor. The oxidation factor (OX) was based upon the recent recommendation of the U.S. EPA, which specify oxidation rates at various stages of landfill gas collection (U.S. EPA, 2015b). Assuming for the first two years prior to implementation of a gas collection system OX = 10%, for years 3-10 OX=20% until the final cover is put in place after 10 years when OX=35%. While this is a revision to the 10% assumption for all years previously used, a recent study (Chanton et al., 2009) estimated OX to range between 22% and 55%. Therefore this range is used for sensitivity analysis.

LFG recovery was based upon the schedule of gas collection efficiency (LCE) presented by Levis and Barlaz (2011). This assumes no gas collection in place for the first two years, 50% collection prior to cell closure at 5 years and then 75% once the cell is closed until after 10 years when the final cover is put in place and the maximum collection efficiency is achieved.

The landfill gas conversion system is modeled based upon default factors provided by the U.S. EPA Landfill Methane Outreach Program (LMOP) to be representative of a typical U.S. LFG project (U.S. EPA, 2015c). The methane that is captured by the LFG recovery system is multiplied by a heat rate conversion factor (Btu/kWh) based upon typical efficiency of the electricity generation system. It is then multiplied by a net capacity factor to adjust for the average load on the generator and takes into account the system availability due to maintenance or repair and the loss of a parasitic load due to operation of onsite equipment. Electricity exported to the grid is assumed to offset U.S. average non-baseload grid emissions. High and low regional grid emissions were analyzed for sensitivity.

A landfill operates primarily through anaerobic decay processes thus it has been argued that a portion of the carbon in the FSC resource that is not degradable through anaerobic processes will remain stored or sequestered in the landfill for greater than 100 years (Barlaz, 1998; Staley and Barlaz, 2009). Barlaz (1998) estimated this portion based upon bio-degradability experiments for several constituents of MSW including MSWFW. This theory was extended to estimate the carbon storage for specific FSCRs based upon the initial carbon content ($C_{initial}$) and the extent of degradation (f_d) as defined by Ebner et al., 2015:

$$f_d = B_o/B_u \quad \text{(Eq. 5-3)}$$

where:

- B_o= the bio-methane potential measured by the BMP assay
- B_u= theoretical methane yield calculated based upon protein, lipid and carbohydrate content as follows (Table 5-4).

and

$$Bu = CP*495 * CL*1016 + Carbo*415 \quad \text{(Eq. 5-4)}$$

- CP= Crude protein (%)
- CL= Crude lipids (%)
- Carbo= % Carbohydrate

A summary of parameters used in the landfill model, their sources and uncertainty ranges are shown in Table 5-5.

Carbon sequestered (CS) was estimated as a percentage of initial carbon as follows:

$$CS_{LF}= C_{initial}*(1-f_d) \quad \text{(Eq. 5-5)}$$

where $C_{initial}$ was estimated from the FSCR resources as follows (Table 5-4).

$$C_{initial} = (CP*0.53 + \%CL*.77 +\%Carbo*0.53) * TS \quad \text{(Eq. 5-6)}$$

Table 5-4: Parameters used to calculate B_u and $C_{initial}$.

Nutrient molar representation $C_nH_aO_bN_c$

Mol Wt.	C	H	O	N
g/mol	12	1	16	14

Nutrient	n	a	b	c	Mol Wt (g/mol)	Bu at STP (mLCH4/g VS)	G C/ gNutrient
Protein	5	7	2	1	113	495	0.53
Carbohydrates	6	10	5		162	415	0.44
Lipids	57	106	6		887	1016	0.77

Table 5-5: Key Parameters used in the landfill treatment pathway

Landfill	Units	Nominal	Low	High	Reference
Diesel use at landfill	L/t	5.83	4	10	U.S. EPA, 2015c
Max oxidation factor (OX)	%	0.35	0.1	0.55	U.S. EPA, 2015c, Chanton et al., 2009[a]
Heat rate of LFG to Electricity conversion	Btu/kWh	11700	10663	13123	U.S. EPA, 2015b
Landfill net capacity Factor	%	0.85	0.8	0.9	U.S. EPA , 2015b
Max landfill capture efficiency	%	0.95	0.85	0.95	Levis and Barlaz, 2011
Lab B_o to L_o Correction factor *(Cf)*	fraction	1	0.7	1	Cho et al., 2012[a]
Methane generation rate (k)	year^{-1}	0.144	0.10	0.229	Levis and Barlaz, 2011

[a]-Used to provide uncertainty range

5.2.4.2 AD treatment pathway

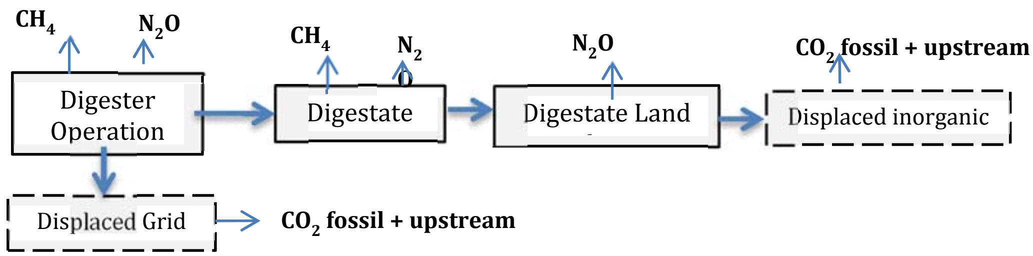


Figure 5-2: System diagram for AD treatment pathway. Dashed lines indicate a system expansion to include displaced processes.

Food waste can be digested as a single substrate or co-digested with other food wastes or manures. Only those impacts associated with the treatment of the food waste were included; for example, no benefits were associated with manure management emission reductions. Biogas produced was assumed to generate electricity, exported to the grid after providing for a parasitic load to operate the digester.

The AD pathway emissions result from digester operation (including displaced grid emissions due to electricity generated), storage of digester effluent (digestate) and land application of the digestate (including displacement of inorganic fertilizer and carbon storage) (Fig. 5-2).

Biogas plant operation was modeled based upon a mesophyllic CSTR biogas plant (Ebner et al., 2015b) with modifications to model specific FSC resources impacts and minor updates to the methodology based upon recent literature.

Bio-methane produced ($QCH_{4,prod}$) was calculated from the bio-methane potential of the FSC resources by applying a methane correction factor (*Cf*) to estimate commercial scale performance based upon lab BMP results (B_o).

$$QCH_{4,prod}=B_o*VS*Cf \quad \text{(Eq. 5-7)}$$

Methane utilized by the generator was estimated by subtracting biogas flared or leaked and electricity generated was estimated by applying a methane conversion efficiency factor calculated from data on a commercial biogas plant performance:

$$QCH_{4utilized} = QCH_{4prod} - QCH_{4leak} - QCH_{4flare} \quad \text{(Eq. 5-8)}$$

$$MWh_{gen} = CE * QCH_{4Utilized}/1000 \quad \text{(Eq. 5-9)}$$

where

- $QCH4_{utilized}$ = Methane utilized (m^3)
- QCH_{4prod} = Methane produced (m^3)
- QCH_{4leak} = Methane losses due to leaks in piping and uncontrolled releases
- QCH_{4flare} = Methane flared (m^3) (emissions due to incomplete combustion when flaring are neglected due to the small magnitude of these emissions; Ebner et al. (2015b)
- MWh_{AD} = Electricity generated (MWh)
- CE = Electricity conversion efficiency (kWh/m^3 CH_4)

Electricity exported to the grid was calculated by subtracting the parasitic load used to operate the biogas plant. Grid offset was calculated based upon the U.S. national average non-baseload emission factor.

Effluent leaving the digester was nominally assumed to be stored in uncovered earthen storage pits until land-applied as organic fertilizer when weather, crop and field conditions allow. During storage residual bio-methane can be released to the atmosphere. Direct and indirect emission factors were used to calculate N_2O emissions during storage. Methane emissions were estimated based upon a residual bio-methane potential factor $B_{o,resid}$ (m^3CH_4/gVS_d) obtained from literature (Ebner et al. 2015b). A volatile solids reduction factor (VS_r) was applied to estimate the VS content of the digestate associated with a given FSC resource as follows:

$$QCH_{4,e} = B_{o,resid} * (VS^* (1\text{-}VS_r)) \qquad \text{(Eq. 5-10)}$$

where

- $QCH_{4,e}$ =methane emissions during effluent storage
- $B_{o,resid}$ = residual bio-methane potential of the effluent
- VS= volatile solids of the FSCR
- VS_r= reduction in volatile solids during digestion

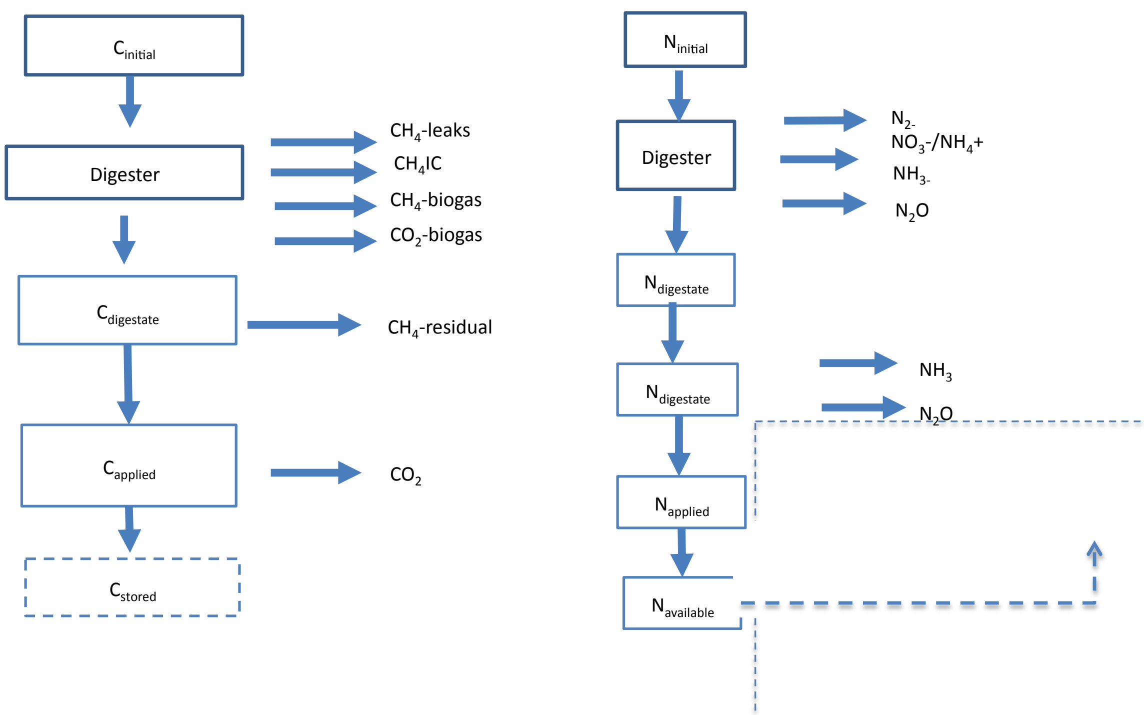


Figure 5-3: Carbon and Nitrogen balance for the AD process. Dashed lines indicate a system expansion to included mineral fertilizer displacement.

When land applied, some of N is volatilized (or is lost through denitrification (N_2 and N_2O), run-off to surface waters (NO_3^-) or leaching to ground water (NO_3^-, NO_2, NH_4^+) (Fig.3). The remaining N has the potential to displace commercial fertilizer. The amount of N readily available for uptake by plants is closely related to the mineral N content. Because commercial fertilizers consist of mineral fertilizer they are considered readily available. The available portion of applied organic fertilizers (i.e., FSC resources, manure, compost, digestate) consists of the mineral N content along with a portion of the organic N that will be mineralized in the near term. An availability factor was used to estimate this portion of total N for a given FSC resource (Poeschl et al., 2012). A single availability factor (based upon MSFW digestate) was used for all resources as the proportion of organic and mineral N is assumed to be similar among FSC resources. However, different factors were used for raw, composted or digested resources, as these processes are known to affect mineral content.

Available N represents the maximum amount of nutrients that can be displaced by inorganic fertilizer. However, due to the imbalance of nutrients provided by the digestate vs. the nutrient demand, some nutrients may be provided in excess of requirements and thus not offset commercial fertilizer. Nutrient management planning is complex and often involves soil analysis, geographic and crop rotation data. A representative example was used to set nutrient demand based upon general fertilizer guidelines for corn crop in New York State, assuming medium soil condition.[42] Phosphorous (P) and potassium (K) demand were determined relative to N demand and compared to nutrients applied (Table 5-6). No offset credit was applied for nutrients (i.e., P or K) provided in excess of nutrient demand.

[42] Cornell University, College of Agriculture and Life Sciences, Fertilizers for corn, web 2015, http://fieldcrops.cals.cornell.edu/corn/fertilizers-corn

Table 5-6: Fertilizer requirements relative to N demand for corn in NYS

Nutrient	kg fertilizer[a]/ha	Availability[c]	Equivalent nutrient/ha[b,c]	Demand relative to N
Nitrogen	80.4	0.65	52.5	1.00
Phosphorous	35.7	1	15.5	0.30
Potassium	35.7	1	29.8	0.57

[a] Average N, P_2O_5 and K_2O requirement for corn estimated based upon http://fieldcrops.cals.cornell.edu/corn/fertilizers-corn

[b] Stoichiometric conversion where mass of P was calculated from P_2O_5 *(162/142), and mass of from K= K_2O *(78/94)

[c] Availability factor or mineralization rate for digestate

Agricultural application of digestate also has the ability to generate long-term carbon storage. There is some evidence to suggest that materials resistant to anaerobic decay (such as lignin) will persist in soil when land applied despite the presence of lignin decaying microorganisms and fungi (Smith et al., 2001). Based on this earlier research, an approach similar to that used in the landfill treatment pathway was considered. However, a relationship between extent of degradation (f_d) and the amount of carbon persisting long term in soils (i.e., soil carbon storage) has not been established. Therefore, the carbon storage factor (% carbon stored/carbon applied) for MSFW digestate obtained from Danish agronomic modeling was applied to the remaining carbon (Hansen et al., 2006).

Key parameters used in the anaerobic digestion module, their sources and uncertainty ranges are summarized in Table 5-7 and select parameters are discussed below.

Table 5-7: Key parameters for the anaerobic digestion treatment pathway, uncertainty range and source.

Anaerobic Digestion	Units	Value	Low	High	Source
Methane correction factor	%	0.9	0.7	1.0	Ebner et al. 2015, Møller et al., 2009[b]
Methane flare	%	0.05	0.03	0.16	Levis and Barlaz, 2011 Gooch et al., 2003[b]
Methane leaks	%	0.025	0	0.1	Ebner et al., 2015[a] IPCC, 2006[b]
Methane incomplete combustion factor	%	0.005			Dressler, 2012
Conversion Efficiency	kWh/m^3 CH_4	4.19	3.2	4.41	Ebner et al, 2015[a]
Parasitic load	%	0.12	0.1	0.2	Ebner et al, 2015
VS_r destruction	%	0.55	0.4	0.7	Ebner et al., 2014
Effluent residual bio-methane potential	m^3CH_4/kg VS	0.054	0.004	0.074	Ebner et al., 2014
Storage direct N_2O emission factor	$kgCO_2e/t$	0.005	0.0025	0.01	IPCC, 2006
Storage indirect N volatilization factor	kgNvol/kg N	0.26	0.05	0.5	IPCC, 2006
Indirect emission factor	kg N_2O-N/kg N				IPCC, 2006
Direct N_2O emission factor	kg N_2O-N/kg N	0.0125	0.005	0.05	IPCC, 2006
LA N volatilization factor	$kgCO_2e/t$	0	0.05	0.5	IPCC, 2006
AD xport to field	km	10	5	30	Ebner et al., 2015
AD N availability factor	$kgCO_2e/kg$ N	0.65	0.4	0.8	Poesch et al., 2012 Boldrin et al., 2009
K availability	% K applied	1			Moller et al., 2009
P availability	% P applied	1			Moller et al., 2009
Carbon storage factor	%	0.1	0.02	0.14	Hansen et al, 2006

[a] Calculated

[b] Determined uncertainty range

5.2.4.3 Compost treatment pathway

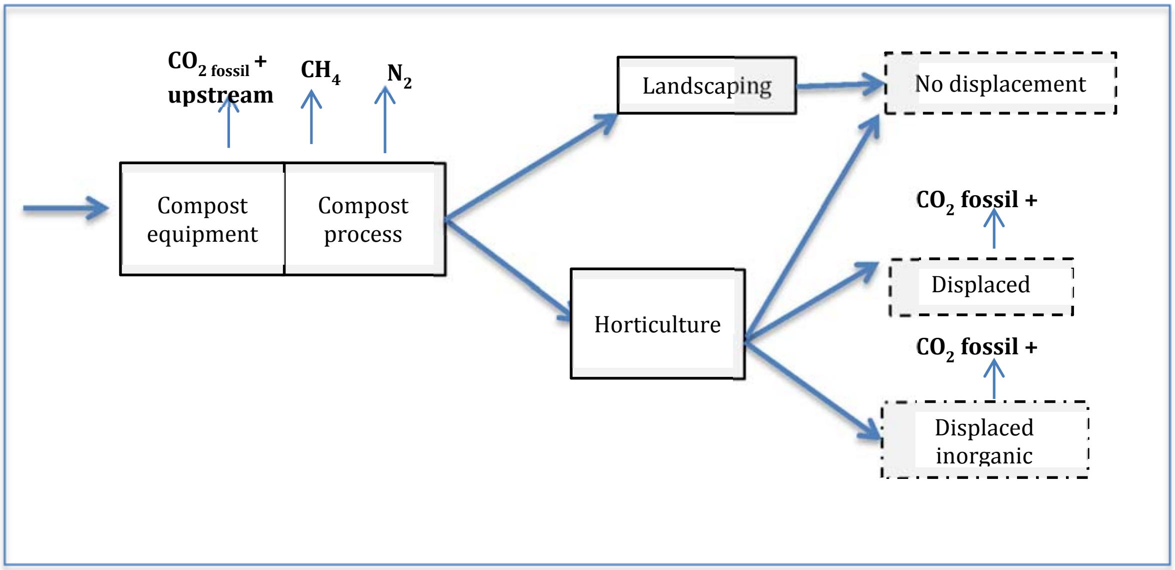


Figure 5-4: Compost treatment pathway system diagram.

The GHG impacts associated with composting consist of those associated with operation of the compost facility, biological decay of the waste and utilization of the compost (including displacement of alternative soil amendments and long term carbon storage as illustrated in Fig. 5-4).

A variety of technologies can be employed in compost facilities; these can broadly be classified either as open or closed technologies. Closed systems generally use more electricity while open processes often use more diesel fuel. Also, some closed facilities can employ biofilters to treat gaseous emissions. However, due to variations in management practices and implementation the ranges of these factors can vary broadly (Boldrin et al, 2009). Therefore, nominal parameters were based upon an open windrow system and parameter ranges were used to account for variability including those related to composting technology.

Mass balances were used to track biological degradation of C and N (Fig.5-5).

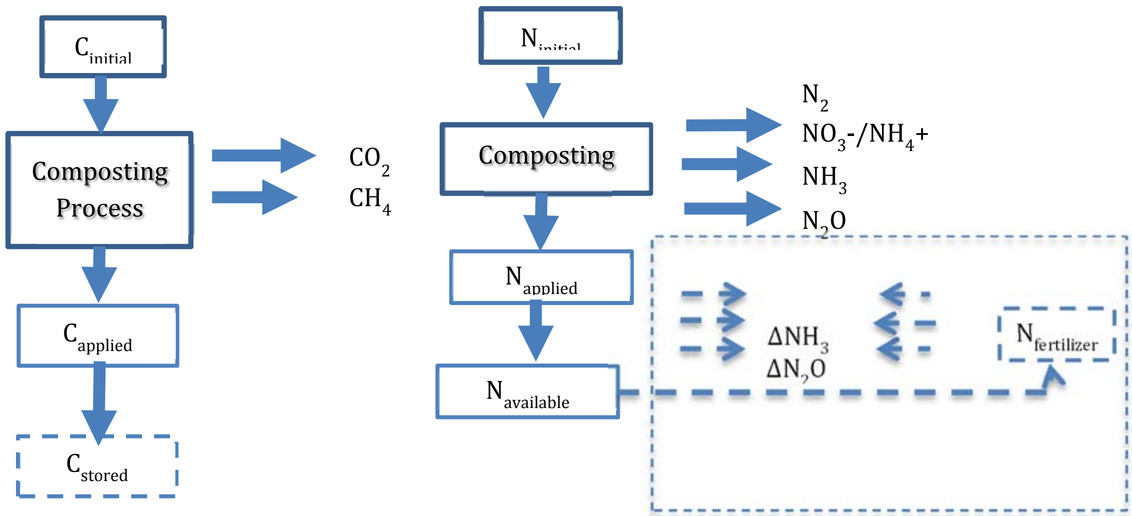


Figure 5-5: Mass balance for C and N in composting. Dashed arrows represent potential system expansions to accommodate difference in N losses relative to mineral fertilizer (agricultural) and/or displacement of mineral fertilizer production and upstream emissions (agricultural or horticultural).

Per the ISO 14041 (2006) and the ILCD Handbook (2010) the system was expanded to include the functionally equivalent alternative products. However, compost can substitute a variety of products and deliver multiple functions (Fig.5-6).

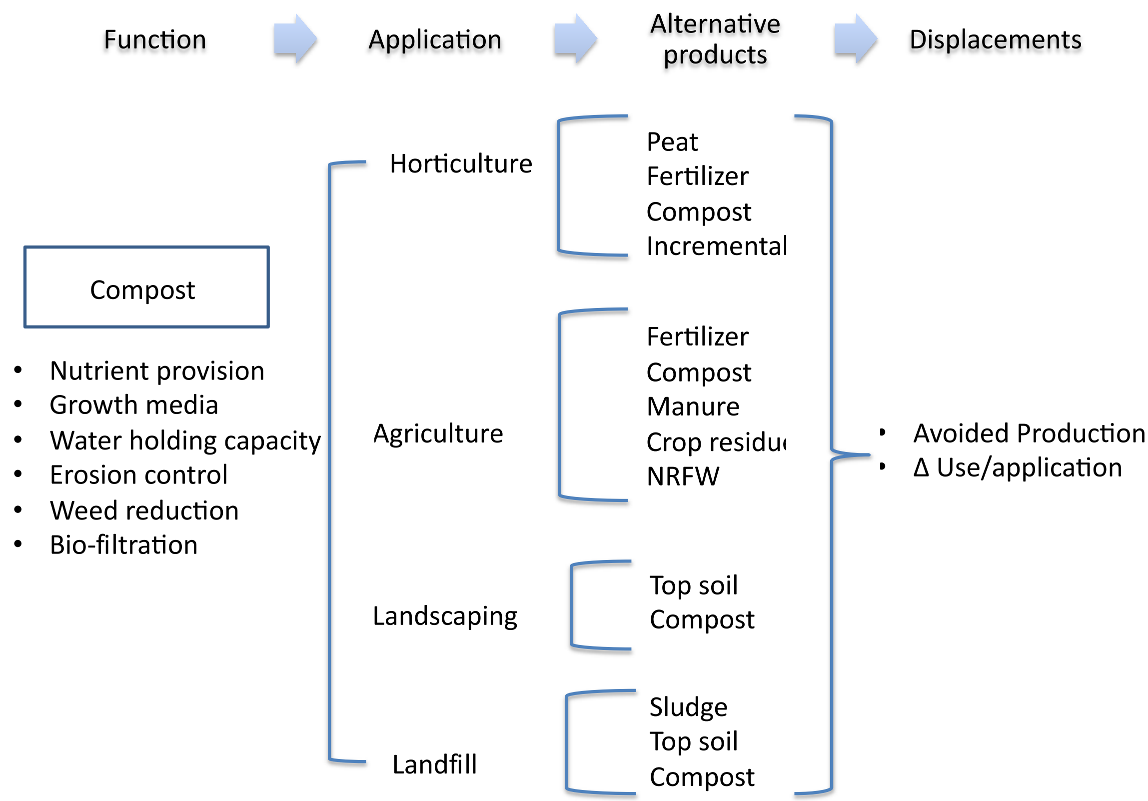


Figure 5-6: Functions, alternative products and displacements for compost pathway

Despite the broad potential for compost application, a Danish study reported that only 41-58% of users of compost from facilities that process MSWFW, claimed that the compost they either received or purchased, substituted an alternative product. The compost was generally used in horticultural activities (residential and municipal). Substituted goods included peat as a growth media (21%), mineral fertilizer (18%) or manure (11%)(Andersen, 2010). While specific market data for the U.S. is not available preliminary research from the U.S. EPA suggest that the most common markets for U.S. compost are also horticulture and landscape applications (U.S. EPA, 2015c). Therefore the system was expanded in a blended scenario to proportionally represent displacement of the functionally equivalent products.

For the portion of compost that displaced peat, the system was expanded to include displacement of materials and emissions associated with extraction and transportation of peat. In addition, since peat is formed over a long time period of time as a result of degradation of plant material under anaerobic conditions the carbon in peat is effectively sequestered; when extracted, peat decomposes releasing CO_2, which is considered a GHG emission as it disturbs the natural peat carbon cycle. No experimental data was found on releases of N_2O from growth media in horticulture, emissions due to use of compost as a growth media were assumed to be equivalent to those of peat. Furthermore, since the long-term fate of the soil is unknown no long-term carbon storage was assumed for use of compost substituting for peat.

As a fertilizer substitute in horticulture, the system was expanded to include the materials and net emissions associated with production and application of displaced equivalent mineral fertilizers. A compost N availability factor was used to estimate plant available N based upon the mineral content of compost. P and K were again assumed to be 100% available. The available nutrients were assumed to perfectly substitute mineral fertilizer in the horticulture case. A variety of commercial fertilizer formulations exist and in reality the horticulturist is unlikely to make perfect substitution of each nutrient so this is considered a best-case scenario.

Substitution of manure in the blended scenario did not avoid any impacts, as it was considered a low value by-product of livestock production (and thus demand was not likely to affect production).

In addition to the blended scenario an agricultural application of compost was also considered in a sensitivity analysis. In this scenario rational application of compost was assumed to displace mineral fertilizer (based upon average demand for corn as described for the AD treatment pathway). Losses due to volatilizes, denitrification, run-off to surface waters or leaching to ground water were accounted for and net direct and indirect N_2O emissions were calculated (Fig. 5-5). Long-term carbon storage impacts (relative to a mineral fertilizer reference scenario) were calculated based upon agronomic modeling for composted MSFW (Bruun et al., 2006, Smith et al., 2001). (As discussed above (regarding AD), a factor relating FSC resource digestibility to carbon storage was not available.)

It is worth noting than an analysis using the U.S. EPAs CENTURY model has generated significantly higher carbon storage estimates for compost. However, the carbon storage mechanisms modeled are not well understood and these results have recently come into question (U.S. EPA, 2015c). The U.S. EPAs WARM model assumes carbon from compost remains stored in the soil through two main mechanisms: soil carbon increases in depleted soils and carbon stored in non-reactive humus compounds. The former is based upon evidence suggesting that organic matter (manure, compost or digestate) steadily applied to soils results in a gradually decreasing, build-up in soil carbon over time; upon termination of the organic matter addition the results gradually reverse (Smith et al., 2001; U.S. EPA, 2015c). Thus it can be inferred that proper and long-term application of organic matter improves soil health especially in cases of depleted soils. However, it cannot be assumed that compost is applied to depleted soils. Furthermore, adding this effect to an LCA of waste management seems to confound two analyses, the first having to do with questions relating to long-term organic vs. conventional agricultural practices and the second relating to impacts associated with marginal substitution of waste managment co-products.   For these reasons this impact was not included at this time and only the latter which as estimated in European models (Hansen et al., 2006, Bruun et al., 2006, Yoshida et al., 2015)

A summary of parameters used in the compost treatment pathway, their sources and uncertainty ranges are shown in Table 5-8.

Table 5-8: Key parameters used in compost treatment pathway, uncertainty range and references.

Compost	Units	Value	Low	High	Reference
Compost operation diesel use	L/t	3	0.13	6	Boldrin, 2009
Compost operation electricity use	KWh/t	0.023	0.023	65	Boldrin, 2009
Carbon degradation	Fraction initial C	0.58	0.4	0.83	Boldrin, 2009
Composting CH_4 emissions	Fraction degraded C	0.02	0.008	0.036	Boldrin, 2009
Composting direct N_2O emissions	Fraction degraded N	0.005	0.001	0.018	deGuardia, 2010
Composting NH_3 emission	Fraction degraded N	0.5	0	0.9	assumed
Composting N loss	Fraction initial N	0.43	0.23	0.57	deGuardia, 2010 Beck Friis, 2000[a]
Compost mass reduction	Fraction initial mass	0.6			Boldrin, 2009
Peat substitution factor	kg compost/kg peat	1	0.2	1	Boldrin, 2009
Compost N availability	Fraction N applied	0.2	0.2	0.4	Boldrin, 2009
Compost land application direct N_2O emission factor	Fraction N	0.034	0.017	0.051	Yoshida et al., 2015
Compost runoff/leaching	Fraction N	0.6765	0.268	1	Yoshida et al., 2016, Hansen et al., 2006[a]
Compost volatilization coefficient	Fraction N	0.016	0.000	0.200	Yoshida et al., 2017, Bruun et al., 2005[a]
Carbon Storage factor	Fraction	0.10	0.02	0.14	Boldrin et al., 2009

[a]Used to determine uncertainty range

5.2.4.4 Animal Feed treatment pathway

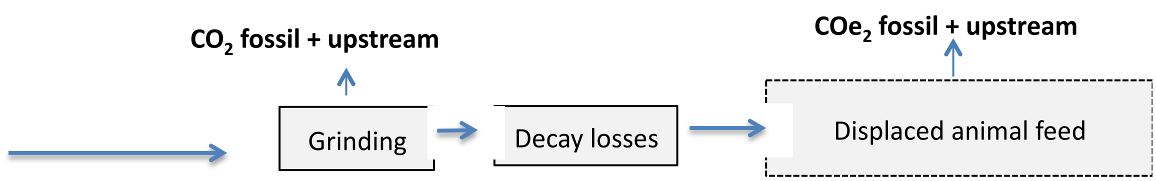


Figure 5-7: Animal feed treatment pathway system diagram.

A variety of FSC resources can be utilized as feed by livestock such as hogs, poultry and cattle. These resources can be diverted to feed animals directly (as wet feed) or after processing which can include drying or reconstituting. Data on emissions associated with food waste diversion to feed animals is limited. A simple wet process was modeled where resources are ground and directly incorporated into feed rations for cows to displace commercial animal feed. Thus the emissions consist of fossil fuel use to grind the resources and an offset for avoidance of the functionally equivalent animal feed. It was assumed that the FSC resource was not perfectly utilized (due to decay or management losses) and a shrinkage factor was applied to estimate the portion of FSC resource that does not substitute commercial animal feed.

Animal feed formulation can be complex, often employing advanced nutritional analyses and software tools. In addition to achieving a balance of minerals and nutrients, integrating FSC resources requires consideration of animal tolerance, which may restrict their use. Management practices and infrastructure may also be required to utilizer FSC resources. However, when FSC resources are used, they generally are sources of energy displacing corn feed in ration formulas. Therefore, a simple displacement was calculated based upon the principal of TDN, which stands for total digestible nutrients. TDN is actually a measure of energy, based upon a carbohydrate equivalent basis. Thus TDN (%DM) is obtained by summing digestible proteins, digestible fat and digestible carbohydrates by applying digestibility coefficients to the compounds resulting from proximate analysis. A distinction is made between readily digestible carbohydrates and less digestible carbohydrates and separate coefficients are provided for each.

Thus TDN can be calculated as follows.

%TDN = %CP (PC) + (2.25(%CL*CLC)) + %NFE (NFEC) + %CF (CFC) (Eq. 5-11)

where:

- %CP= crude protein content
- PC= protein digestability coefficient
- %CL = crude fat content (lipids or ether extract)
- CLC = crude lipids digestibility coefficient
- %CF= crude fiber content. This is intended to represents insoluble carbohydrates or less digestible carbohydrates.

Note: CF has been phased out as a parameter in feeds for ruminants due to underestimates especially for forages where lignin content is substantial. Most formulations have replaced a term based upon acid detergent fiber (ADF).

CFC= crude fiber digestibility coefficient

%NFE= nitrogen free extract, sometimes referred to as non-fiber carbohydrate. This represent soluble carbohydrates (such as starch or sugar) that are readily digestible. %NFE is determined by deducting measured proximate factors such that

$$\% \text{ NFE} = \% \text{ DM} - (\% \text{ EE} + \% \text{ CP} + \% \text{ ash} + \% \text{ CF}) \qquad \text{(Eq. 5-11)}$$

- NFEC= nitrogen fiber extract digestibility coefficient

TDN data was not available for all FSC resources. Therefore, this formula was modified to estimate TDN based upon the key resources characteristics.

Such that modified TDN was:

%TDN = %CP (PC) + (2.25(%CL*CLC)) + %Carbo (CC)) (Eq. 5-11)

where:

- CC= carbohydrate coefficient

The digestibility coefficients (PC and CC) were applied to the percent crude protein and percent crude lipids respectively. To estimate carbohydrate digestibility, resources were classified as containing predominately highly degradable carbohydrates, medium carbohydrate mix or predominately less degradable carbohydrates based upon their extent of degradation (*fd*) (Table 5-9).

Table 5-9: Parameters used in modified TDN calculation

	f_d	f_d <60%	f_d >95%
Protein digestability coefficient (PC)	0.85		
Lipid digestibility coefficient (LC)	0.8		
Carbodigestability coefficient (CC)	0.74	0.6	0.9

The model was then calibrated using TDN references from the literature (Table 5-10).

Table 5-10: Calculated modified TDN compared to TDN for several FSC resources and sources.

	Modified %TDN	Bath et al., 1995	BEEF magazine, 2015	NRC, 2012
MSWFW	88%	80%[a]	80%	
Whey	81%	81%		81%
Tomato Pomace	69%	63%	64%	
Apple Pomace	67%	69%	70%	
Brewers spent grains	85%	66%	85%	73%
Baked goods	85%	89%	90%	89%
Canned goods	71%	72%		
Coffee grounds and filter paper	73%		20%[b]	
Fresh produce	82%			
Post consumer	95%			
Prep waste	56%			
Sweet Cereals	74%			
Salad	63%	51%		
Dry goods	75%		76%	
Refrigerated and frozen goods	92%			

[a] Garbage, municipal cooked
[b] Coffee grounds

The resource's equivalent energy as determined by TDN was then used to calculate a displacement factor (DF) for corn feed (kgFSC resource/kgCorn feed) as follows:

$$DF_i = \frac{TS_i * TDN_i}{TS_c * TDN_c} \quad \text{(Eq. 5-12)}$$

where:

- TS_i= total solids content of $resource_i$
- TDN_i=modified TDN of $resource_i$
- TS_c=solids content of corn feed (88%)

- TDN_c= TDN of corn feed (88%)

Displaced emissions were calculated by applying an emission factor for cultivation and production of corn animal feed to the amount of corn feed displaced.

Key parameters used in the animal feed treatment pathway, uncertainty ranges and sources are shown in Table 5-11.

Table 5-11: Key parameters used in the Animal Feed utilization pathway

Animal Feed	Units	Value	Low	High	Source
AF shrinkage	%	0.1	0.05	0.5	
Corn feed emission factor	$kgCO_2e/kg$	-592			Weidema et al., 2013

5.3. *Results*

Net GWP impacts vary across FSC resources and treatment pathways (Fig 6-7). Net impacts for landfill treatment resulted in the highest GWP impacts for all FSCRs. The animal feed pathway was net negative for all resources and the preferred pathway all except salad. The next preferred pathway shifted depending upon resource characteristics, with AD the having lower net emissions for MSWFW and half of the resources and compost preferable for the other half. AD treatment had a net negative impact for many resources (except apple pomace, tomato pomace, prep waste and salad). Compost treatment pathway results were net negative for canned goods, fresh produce, tomato pomace, apple pomace, whey, prep waste and salad. (Fig. 5-8).

AD treatment had a net negative impact for many resources (except apple pomace, tomato pomace, prep waste and salad). Compost treatment pathway results were net negative for canned goods, fresh produce, tomato pomace, apple pomace, whey, prep waste and salad. (Fig. 5-8).

The landfill pathway showed the highest range varying from 3115 kgCO_2e/t baked goods and 111kgCO_2e/t salad mix (with results for MSFW 623kgCO2/t). AD treatment pathway results ranged (282) kgCO_2e/t baked goods to a positive impact of 8kgCO_2e/t apple pomace (with (31) kgCO_2e/t MSFW) and ranged from (61) kgCO_2e/t salad mix to 156kgCO_2e/t baked goods with MSFW impacts of 14.7kgCO_2e/t

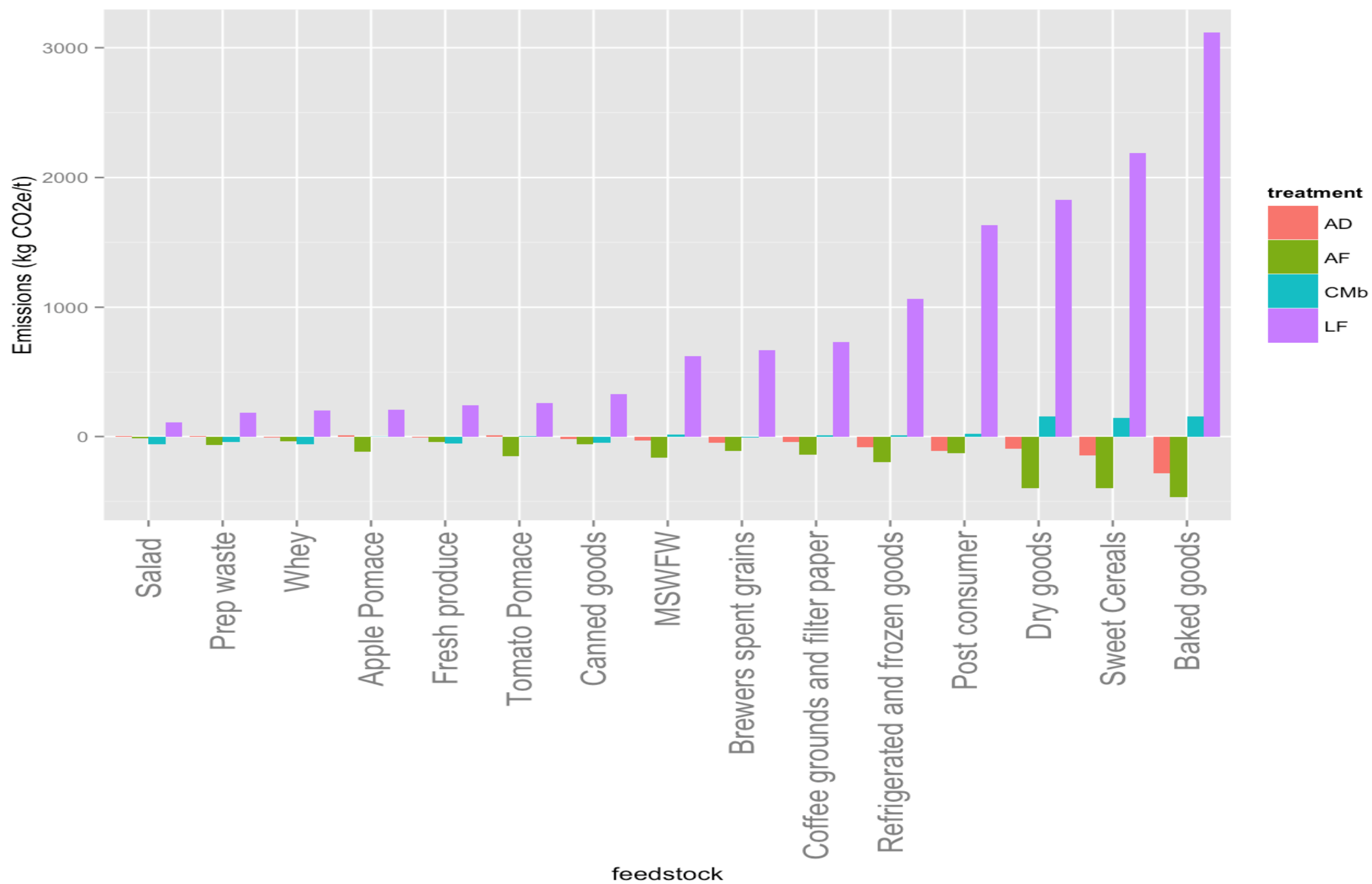


Figure 5-8: Net climate change impact (kgCO_2e/t resource). For AD= AD treatment pathway (with baseline fertilizer displacement scenario); AF= Animal feed pathway; CM=compost treatment pathway (with blended displacement scenario); LF= Landfill treatment pathway

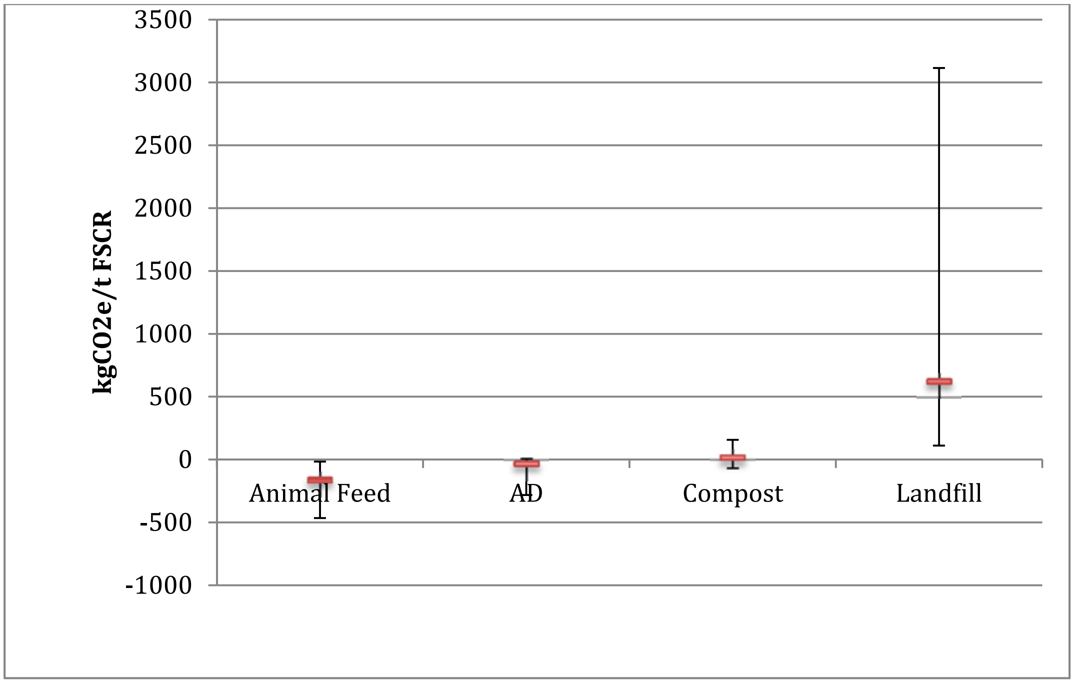


Figure 5-9: Range of GHG impacts for the four treatment pathways. Red bar represents nominal, error bars represent the range based upon resources characteristics.

5.3.1 Net impact of diversion:

When considering FSC resource management it is important to realize that treatment via a given pathway implies diversion (and thus avoidance) of alternative pathways (Ebner et al., 2014). In all cases diversion of resources from landfill to either animal feed or compost offered the largest benefit, ranging from (3581kgCO_2e/t baked goods) to (173kgCO_2e/t salad) (Table 5-12). FSC resources with lower solids content and bio-degradability were least sensitive to pathway (i.e., the maximum impact for salad was 173kgCO_2e/t) while those with high solids content and high degradability were the most sensitive (i.e., 3581 kgCO_2e/t baked goods).

Table 5-12: Climate change impact (kgCO2e/t) for FSC resources and treatment pathways. Maximum net impact of diversion (kgCO2e/t) and maximum net impact diversion pathway (from- to)

FSC resources	AF	AD	Compost	Landfill	Max net impact	Recommended diversion
Salad	-16	3	-61	111	173	LF to CM
Prep waste	-55	5	-47	203	258	LF to AF
Whey	-36	-7	-59	207	265	LF to CM
Fresh produce	-44	-9	-55	246	301	LF to CM
Apple Pomace	-118	9	-2	208	326	LF to AF
Canned goods	-60	-17	-46	333	393	LF to AF
Tomato Pomace	-153	7	-5	264	417	LF to AF
Brewers spent grains	-110	-47	-6	668	778	LF to AF
MSWFW	-161	-32	15	623	785	LF to AF
Coffee grounds and filter paper	-139	-43	9	730	869	LF to AF
Refrigerated and frozen goods	-196	-82	9	1064	1260	LF to AF
Post consumer	-276	-149	68	1587	1863	LF to AF
Dry goods	-401	-93	156	1824	2225	LF to AF
Sweet Cereals	-396	-144	149	2188	2585	LF to AF
Baked goods	-466	-283	157	3115	3581	LF to AF

5.3.2 Simplified regression estimation

The treatment pathway models consist primarily of successive combinations of linear (with the exception of the first order decay model) expressions involving FSCR characteristics, inventory parameters and emission factors. Thus these models could be simplified mathematically or as in the approach taken here via linear regression of the model outputs (Table 5-13). The resultant linear models provide a simple way to estimates the GWP impact given FSCR characteristics (either measured or estimated). Since the regressions are just another representation of the model (and not a fit to empirical data) perfect fits could be obtained by including the relevant parameters (Table

5-13). (Fit curves and regression statistics are shown in the Appendix.) The intercepts represent factors that are independent of FSC resource characteristics (i.e., purely a function of the mass processed) and the coefficients indicate the impact of resource characteristics.

Table 5-13: Linear estimation formula for treatment pathways

Perfect fit linear model	Scaled linear model
EM_{LF}=18.4 + 7.6 L_o -3.7 $C_{initial*}(1-f_d)$	EM_{LF}=90.2 + 3738.0 SL_o -790.5 $S(C_{initial}*(1-f_d))$
EM_{CM}= -72.4 +2.7 $N_{initial}$ + 0.44 $C_{initial}$	EM_{CM}= -64.0 +46.9 $SN_{initial}$ + 182.6 $SC_{initial}$
EM_{AD}=8.1 – 1.4 Lo + 461.8 * TVS +0.0TKN -0.37$C_{Initial}$	EM_{AD}=1.1 -545.6SLo + 401.5 STVS +20.4STKN – 154.4 $SC_{Initial}$
EM_{AF}=-688*(TS*TDN)	EM_{AF}=-16.2* (-449.5*S(TS*TDN))

In order to gain insight into the relative impact of resource characteristics independent of their absolute values, the variables of the regressions were scaled to the range of values (such that the maximum value for a given resource characteristic had the value of 1 and the minimum value 0). In this case the intercept represents the impact that is independent of FSC resources characteristics as well as the impacts for the minimum values of each resource characteristic.

In the case of landfill emissions, the bio-methane yield (m^3CH_4/t) of the FSC resource was highly correlated to net emissions. While carbon storage (due to the combine influence of initial carbon and extent of degradation) had an offsetting impact it carried only 1/5 the magnitude. Thus while a net negative impact is possible (i.e., for a feedstock with low bio-methane potential and high initial C and extent of degradation) it is unlikely in the range of values for food waste as the sample substrates demonstrated.

The compost pathway showed a negative intercept due primarily to the significant peat offset factored into the blended substitution case. Both initial C and initial N correlated to positive emissions indicating that process emissions (N_2O and CH_4) outweigh carbon storage or fertilizer offsets under nominal conditions. Since both C and N content correlate with solids content a reasonable fit ($R^2 > 0.98$) could be obtained for a regression based only on TS content (given by the expression: EM_{CM}=-70.32 + 242.0)

Bio-methane yield per t resource showed the largest influence on net AD impacts with a negative coefficient indicating that gird electricity offset is greater than leak impacts. However, variation in volatile solids content had nearly as large an impact due to its relationship to residual methane production for stored effluent (assuming a fixed VS reduction). The small positive coefficient for TKN indicates that fertilizer displacement did not offset nitrous emissions related to land application nominally. However, the negative coefficient for long-term carbon storage will likely result in net negative impacts for digestate land application.

The coefficient in the animal feed pathway indicates that the impact per ton of resource (ie. grinding emissions) is not nearly as significant as the solids content and nutritional content.

(Simplified models purely based upon total solids content were calculated for all pathways and are provided in the Appendix, showing R^2 ranging from 0.65 to 0.98.)

5.3.3 Uncertainty and variability

Bernstad et al. (2012) reviewed 25 LCAs of different food waste treatment alternatives (including compost, AD, landfill and incineration) and observed wide variation in results for climate change impact. All of the studies reviewed concerned household food waste or MSWFW and the results obtained in this study fall within the ranges of those reported. While Bernstad et al. (2012) identified the influence of food waste characteristics as one source of variability, system boundary settings and methodological choices where also attributed to causes in variation among studies. Therefore particular care has been taken to ensure consistent system boundaries across all pathways in this study. Additionally, effort has been made to clearly explain methodological choices and to capture uncertainty in parameters through ranges of parameter values.

A Monte Carlo analysis was performed using 1001 samples taken from uniform distributions for each parameter within the range of values (Tables 5-3, 5-5, 5-7, 5-8 and 5-10). The results are shown in Fig. 5-10.

The uncertainty ranges for many of the resources overlap across treatment pathways. Landfill treatment while clearly least preferred for many resources showed overlapping uncertainty ranges for low solids content and low bio-degradable substrates (i.e., apple pomace, tomato pomace, prep waste and salad).

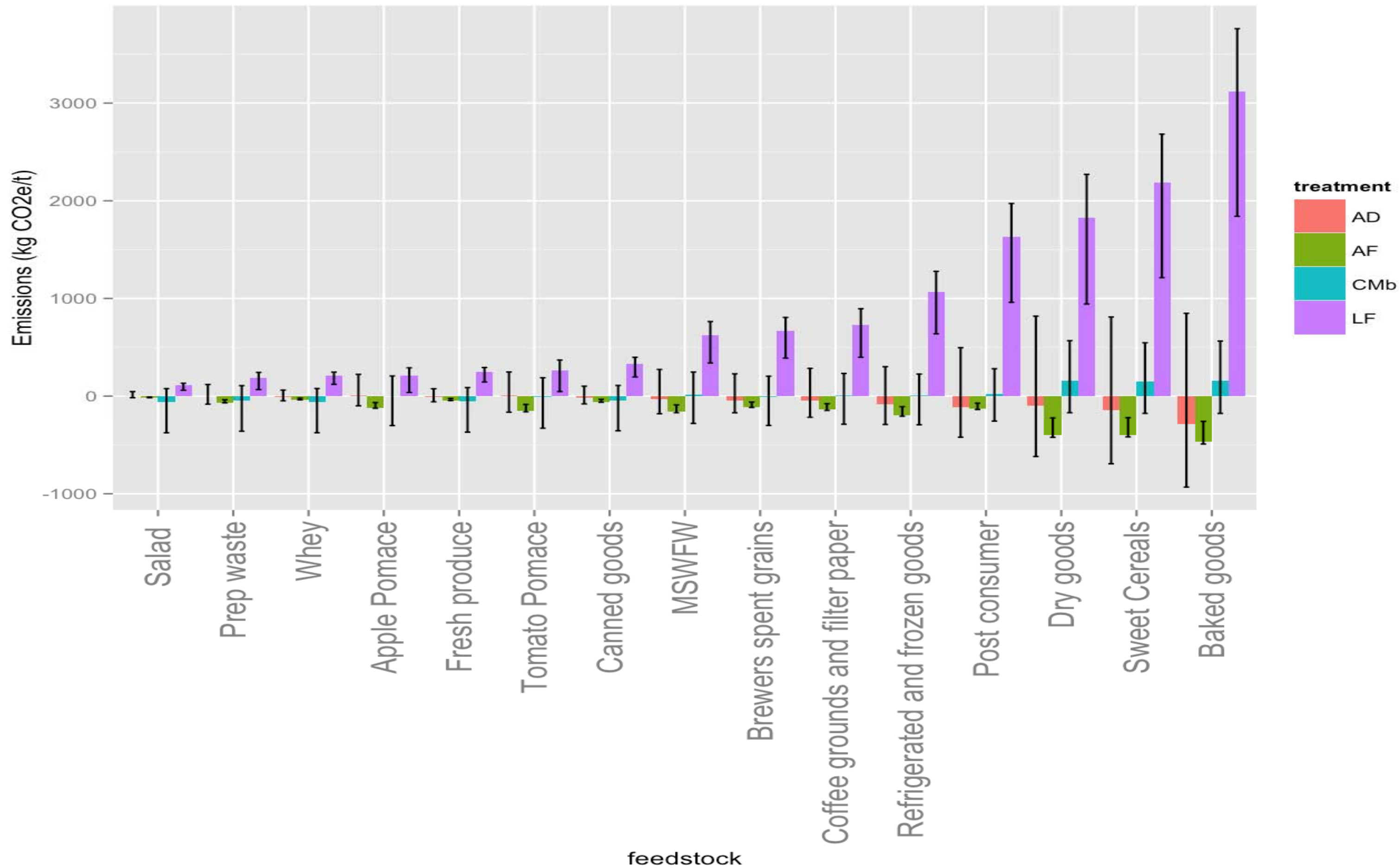


Figure 5-10: Net GWP impacts (kgCo2e/t) for each resource and each treatment pathway. Error bars indicate uncertainty ranges obtained by Monte Carlo analysis.

Therefore a one-dimensional sensitivity analysis was performed, where the maximum and minimum values within the ranges of each parameter were calculated, in order to gain additional insight into and reduce uncertainty (Appendix Table A.2).

The results for the landfill pathway showed that uncertainty in the methane generation rate constant (*k*) had the largest impact on results, where a higher decay rate led to higher methane releases and therefore higher emissions. As mentioned earlier, k is a function both of climate and FSC resource characteristics. Moisture and temperature impact decay rate as does substrate composition. Therefore, rapidly degradable substrates disposed of in landfills in hot, wet climates will result in emissions toward the higher end of the range while those that are slower to degrade and located in colder dryer climates will be toward the lower end. The magnitude of the impact correlated with bio-methane yield (L_o), such that resources with L_o had higher uncertainty. The relative impact of a 30% uncertainty in estimating L_0 (bio-methane correction factor) was similar to that of *k*. Uncertainty in the maximum oxidation factor had slightly less impact. Grid mix variability had a relatively small impact (about 1/6th that of *k*). These factors showed a similar interaction with L_o, resulting in higher uncertainty in FSC resources with higher L_o.

Variability in compost end use assumptions had the largest impact on compost net emissions. Fig. 6-11 below shows the net impact of four compost use scenarios: no displacement, the baseline blended scenario, 100% fertilizer displacement 100% land applied fertilizer displacement, 100% peat displacement.

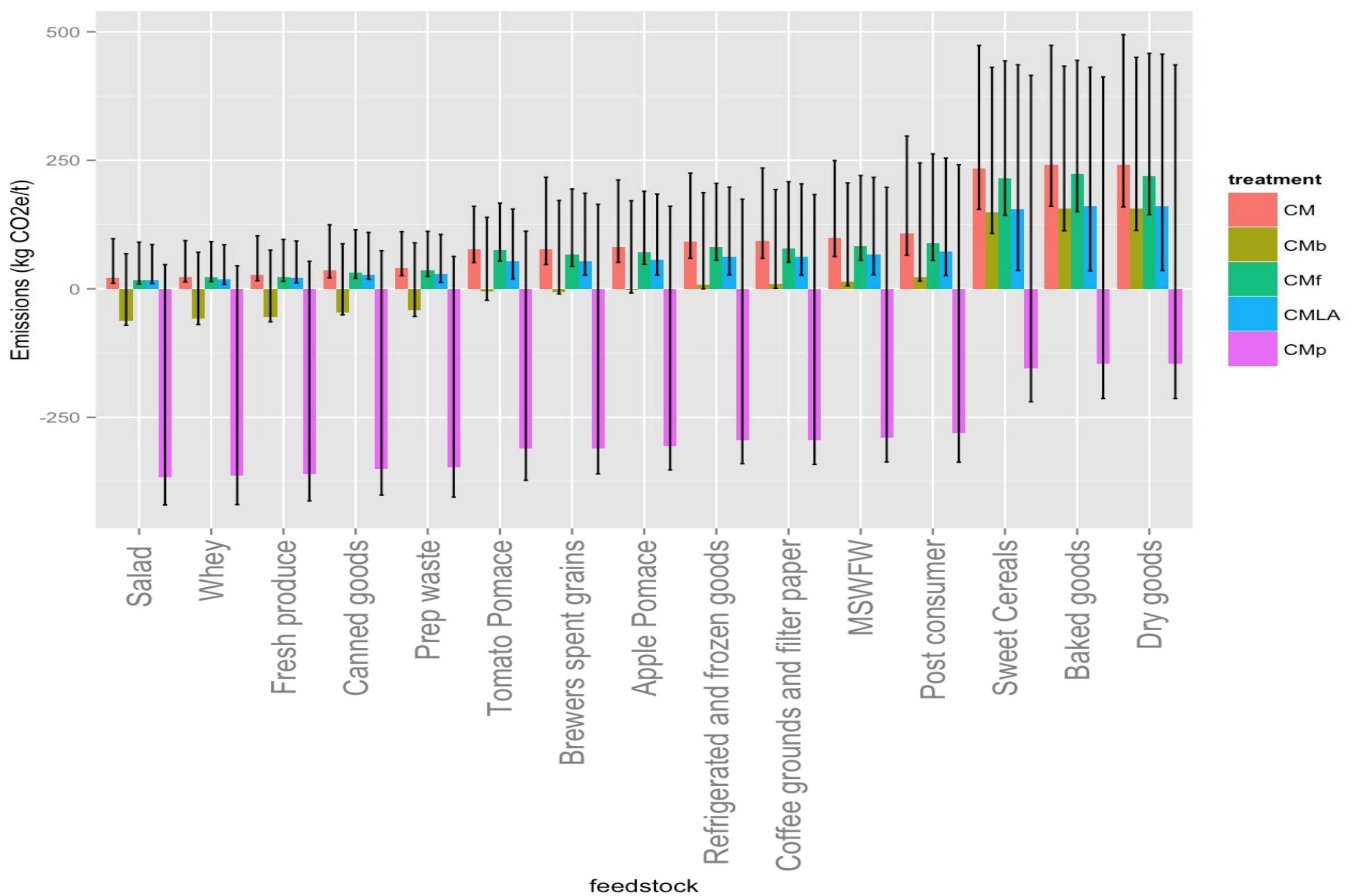


Figure 5-11: Compost end use scenarios. CM=no displacement due to end use; CMb=the baseline blended scenario (21% peat displacement, 18% fertilizer displacement); CMf=compost used in horticulture to displace mineral fertilizer; CMLA=Compost applied to land to displace mineral fertilizer and provide long term carbon storage; CMp=Compost used in horticulture to displace peat. Error bars indicate uncertainty ranges resulting from Monte Carlo Analysis.

The largest contribution to the "no displacement" scenario was biological emissions with fossil fuel use constituting a relatively small contribution. Biological emissions were proportional to FSC resources characteristics with high carbon content and to a lesser extent high N content (i.e., baked goods, sweet cereals, dry goods), leading to higher emissions. In the "fertilizer displacement" scenario the maximum displacement of fossil fertilizer provides an offset to emissions. Where the "land applied fertilizer displacement" scenario provides a smaller fertilizer offset but an additional benefit due to long term carbon storage. However, given the range analyzed net emissions for this scenario are still positive for all resources. The large avoidance of peat production and use in the "peat displacement scenario" results in negative emissions for all resources. Remaining uncertainty in the scenarios is largely due to uncertainty in estimating biological emissions. (Consistent with Bernstadt et al. (2012), fossil fuel use assumptions were not a significant factor). Therefore care should be taken to manage biological emission especially in cases of high C and N content FSC resources.

Uncertainty in the AD treatment pathway was generally large and related to FSC resource parameters. In all but those resources with very low bio-methane potentials (ie. Apple and tomato pomaces), digester leaks had the greatest impact. As this can be controlled in carefully run processes it is an important parameter to monitor. The next most significant source of uncertainty related to the parameters involved in estimating residual methane released during digestate storage (i.e., volatile solids reduction and residual methane potential). Residual methane emissions increase with higher organic loads (i.e., resources with higher TVS). However, they can be limited by proper process controls (i.e., appropriate hydraulic retention times) and management conditions (i.e., brief or covered storage especially in regions with high temperatures). Controlling these factors can have a significant impact and are therefore recommended especially for resources with medium to high organic loads. Controlling these factors can have a significant impact and are therefore recommended especially for resources with medium to high organic loads.

The remaining uncertainty was attributed to several sources. In most cases the largest source were operational parameters (i.e., digester capacity factor, conversion efficiency, percent of gas flared and parasitic load assumptions). Carbon storage factor assumptions also introduced uncertainty, which was amplified by C content of the resource. Land application and fertilizer displacement parameters, while largely influenced by management practices (such as fertilizer application method, nutrient management) and climate had a relatively small impact on uncertainty.

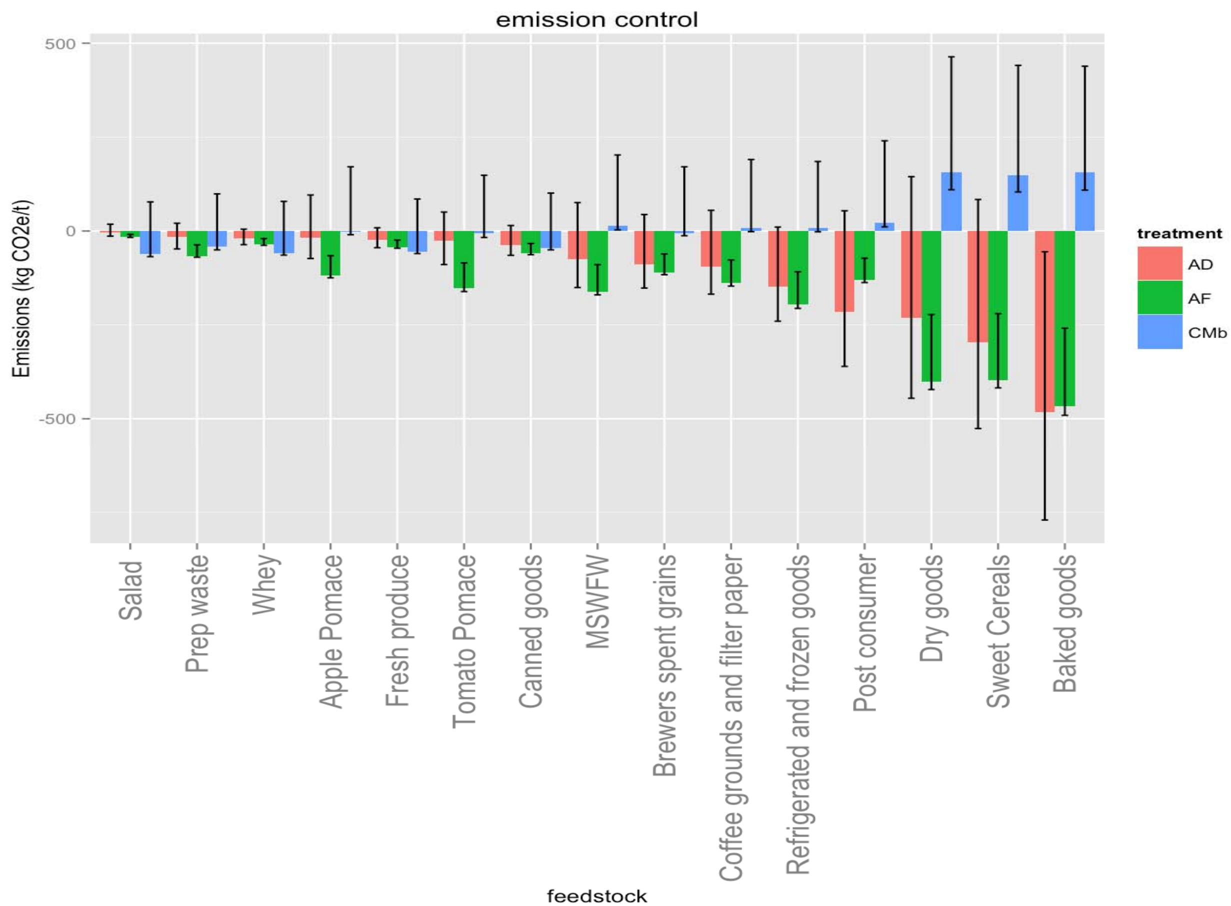


Figure 5-12: "Reasonable case" scenario. Net emissions (kgCO2e/t) and uncertainty ranges resulting from Monte Carlo analysis. AD uncertainty range is based upon fixed nominal assumptions for methane leaks, residual methane production and bio-methane production correction factor in the AD case. Compost uncertainty range assumes the blended end use scenario and nominal value for CH4 emissions.

A reasonable-case scenario was created to provide insight into FSC resource treatment pathway impacts given a well-controlled facility where controllable biological emissions are managed. In this scenario methane leaks and residual methane production were fixed at nominal values for the AD case. Methane emissions were fixed at the nominal values for the compost case and the blended displacement scenario was assumed. The results still show overlapping uncertainty ranges indicating that for well operated facilities AD and compost treatment pathways generate comparable emissions for most resources, while animal feed is clearly preferred to compost for high solid resources.

5.3.4 Conclusions

The ORACAS model was used to calculate emissions for various treatment pathways using FSC resources common to NYS to demonstrate the impact of resource characteristics on net climate change impacts. The results showed that estimating the impacts of FSCR treatment based upon the characteristics of MSFW may result in significant error. The impact was generally greatest for highly bio-degradable resources for landfill and AD. Compost and animal feed showed less of variation and a correlation to solids content. Linear models were provided to estimate net emissions and gain insight into the impacts of FSC resource characteristics. A Monte Carlo analysis was run to analyze uncertainty. Uncertainty in results was related to variability, parameter uncertainty and modeling assumptions some of which interact with resource parameters. The greatest impact on uncertainty of landfill emissions related to methane production rate constant (k). Rapidly decaying resources in hot, wet climates will have the largest magnitude of impact. Assumptions concerning compost end use had the largest impact on compost uncertainty. Using compost productively to displace horticultural or agricultural products can reduce net emissions. Management of leaks and digestate had the largest potential to reduce AD uncertainty especially with treatment of resources with a high bio-methane potential. The ORCA model provides a useful tool to estimate GWP impacts based upon a set of resource characteristics that can be modified based upon changing assumptions or incorporated into more complex models.

Chapter 6 Conclusions and recommendations for future work

The food supply chain (FCS) generates resources at every stage, from "farm-to-fork". Management of these resources can have broad social, environmental and economic impacts. This issue is of particular relevance in New York State, due to several current trends. The first is the movement nationally and locally, to increase distributed and renewable energy generation, which has been driven by concerns over energy independence, resiliency and resource scarcity. The second, motivated by climate change and land use concerns, is the recent landfill restrictions on commercial food in NYC and several Northeastern States. Lastly, the increasing importance of the State's agriculture and food processing industries which has driven increasing measures to support efficiency and growth in this sector. These three trends make management of FSC resources one of New York State's most pressing sustainability issues. The transition from waste to resources offers many opportunities but also raises many questions. This dissertation has addressed several of these questions in support of increasing the sustainable utilization of food supply chain resources in New York State.

In Chapter 2, the language and framework for analysis was provided. Food supply chain resources were defined to include both high and low solids content, edible and non-edible, resources leaving the food supply chain, which spans post-harvest through consumption. Estimates of resources generated at each stage of the supply chain were provided and the resources were characterized. A set of FSC resource generation factors was developed based upon recent literature and datasets. Based upon data collected from 97 food processors, resources from this sector were also characterized. In a cradle-to-grave approach, utilization pathways were also quantified and the flow of resources to utilization pathways was reported. Finally, geographic information was also provided on key resource generators and utilization pathways.

The results showed that the food processing and consumption phases generated the most resources. The most significant resources generated included fruit and vegetable processing waste, whey, brewery waste, bakery waste and commercial food waste. A variety of utilization pathways were reported, including donation to food banks for human consumption, diversion to feed animals, composting and anaerobic digestion. Despite several limitations noted, this analysis provided a foundation for future work.

In Chapter 3, information was provided on the climate change impacts of two utilization pathways: anaerobic co-digestion (AcoD) and waste-to-ethanol via fermentation. Both of the processes selected were operating in NYS at the time and both were novel relative to applications elsewhere. Since most utilization pathways have the potential to generate biological emission of powerful GHGs (i.e., CH_4, N_2O and CO_2) information on climate change is one important criterion to help evaluate and inform sustainable FSC resource utilization. Individual results, contributions and limitations were presented for each process in Chapter 3. However, both processes showed favorable results, with net negative emissions impact relative to alternative processes. A significant factor identified in both studies was the effect of avoided processes. Of particular relevance is the fact that FSC resources must follow a utilization pathway (either actively or passively) and treatment by one pathway implies avoidance of another. Thus, one of the main conclusions of this work is that utilization pathways must always be evaluated in a relative context. The impact of a given utilization pathway can only be assessed if the impacts of the alternative pathways are known as well.

Both the anaerobic digestion and fermentation processes utilized common FSC resources available in New York State. The waste-to-ethanol process utilized retail and food processing resources (beverage/syrup/sauce sector). The AcoD process utilized dairy manure and a combination of dairy processing waste, GTW/DAF and other food resources. The results presented were based on primary data obtained from operating facilities converting these actual FSCs. In the AcoD case in particular, some general conclusions can be extended to similar dairy manure and FSC resource co-digestion facilities. For example, the displacement of grid electricity provides a significant offset of greenhouse gas emissions. Controlling fugitive emissions and residual digestate storage emissions are significant opportunities to reduce net environmental impact. However, the specific FSC resources utilized will impact system performance as well as the impacts of alternative utilization pathways for that resource, which as explained above must also be considered. Extension of these results to predict the impacts of utilizing other FSC resource are discussed further below in relation to Chapter 5.

In Chapter 4, the anaerobic digestion of commercial FSC resources was studied. Combining estimated retail sector and food service (or “out-of-home consumption”, comprised of institutions, entertainment, lodging and restaurants) results in approximately 1.3 million tons of FSC resources produced annually in New York State. These commercial food wastes are often the initial targets of mandatory recycling laws such as the recent legislation in NYC. Anaerobic digestion of these resources is one emerging option as is anaerobic co-digestion (currently utilized by several supermarket stores in the Finger Lakes region). This study characterized 22 source separated substrates and co-digestion blends and conducted BMP testing to determine key bio-methane parameters. The bio-methane potentials of commercial resources ranged from 165 to 496 mL CH_4/g VS. Substrates high in lipids or readily degradable carbohydrates showed the highest methane production. Bio-methane potential of co-digested substrates showed a slight synergistic bias (-5% to +20%) on average, relative to the weighted bio-methane potential of the individual substrates. Apparent hydrolysis rate coefficients ranged from 0.19 d^{-1} to 0.65 d^{-1}. One of the novel contributions of this work was the development of a co-digestion rate index to compare the apparent hydrolysis rate of co-digestion blends with that predicted by combining individual substrates. The combined substrates demonstrated an increase in the rate of apparent hydrolysis. This could be important, as it may lead to shorter hydraulic retention times and improved digester performance. These parameters are important to advance modeling of AcoD system, and further expand utilization of commercial food waste in anaerobic digestion.

In Chapter 5 the results of the BMP tests in Chapter 4 and the LCAs in Chapter 3 were combined to estimate the climate change impact of the utilization pathways identified in Chapter 1, based upon individual FSC resource characteristics. The Organic Resource Climate Assessment Simulator (ORCAS) model was developed as the main outcome of this effort. The results showed that FSC resource characteristics can have a significant impact on treatment pathways. The largest impact was observed for highly biodegradable FSC resources in the landfill and anaerobic digestion pathways. Compost and animal feed utilization pathways showed a correlation to solids content, but the variation was smaller. Linear models were also provided to estimate net emissions based upon FSC resource characteristics. Uncertainty related to modeling assumptions and parameter estimation was significant, and quantified using a Monte Carlo analysis. Using compost productively to displace horticultural or agricultural products can reduce net emissions. Management of leaks and digestate had the largest potential to reduce AD uncertainty, especially with treatment of FSC resources with a high bio-methane potential.

Opportunities for future work have been discussed throughout this dissertation, however several significant opportunities are highlighted below:

- **Coordination and improvement in FSC resources data.** Collecting data to analyze the FSC resources was tedious, and many gaps and inconsistencies were identified. A coordinated, streamlined and repeatable process should be implemented. Data should be collected from establishments that conduct audits or maintain diversion programs to inform FSC resource generation. Additionally, research should be conducted periodically to audit facilities. The highest priority should be to update and provide additional data on the hospitality, entertainment and retail sectors (including coffee and ice cream shops).
- **Communication to promote resource utilization and remove barriers.** An organization should be charged with the function of disseminating relevant information. This includes sharing data on generators and utilizers through the Organic Resource Locator (ORL) and other mechanism, communicating relevant information (i.e., information on liability protection and donation, etc.) and removing barriers (i.e., coordinating technology development, etc.).

- **Further research into FSC resources not included in this analysis.** The most notable resource that was not quantified in Chapter 1 are fats, oils and greases (FOG), and in particular GTW and DAF. These resources were estimated to have a significant environmental impact (Chapter 3) when sent to a WWTP/Landfill, while diversion to AD was estimated to provide a significant benefit. This impact of the current pathways and potential of alternatives should be thoroughly quantified. Additionally, a plan to utilize these resources should be analyzed considering both additional facilities and the considerable resources in place at POTW.
- **Research and development into valorization of significant FSC resources**. Several resources were identified to be generated in significant quantity (Chapter 1), including brewery waste, commercial dairy waste, bakery waste and retail waste. Opportunities to utilize these resources include production of secondary food products such as protein powders and nutraceuticals, bioplastics, industrial alcohols and chemicals, and other waste-to-energy methods, especially thermochemical methods such as gasification and pyrolysis which may be well suited for low moisture content materials. Drying and de-packaging technologies should also be explored as a means of further expanding the available FCS resources available for valorization. Research and development as well as implementation support are particularly important for mid-size generator which may have a high impact in aggregate, but not have the resources to act individually.
- **Research on methane emissions in AD**. The environmental analysis of AD highlighted the impact of digestate residual methane emissions (Chapter 3), yet this finding was based largely upon prior European studies. Because feedstock, climate

and management practices influence emissions, this should be evaluated and a factor provided to estimate residual methane emissions. Similarly, data on fugitive emissions was based upon European data prior to regulation in the EU. These emission should be measured at NYS biogas plants.

- **Development of AcoD**. AD on-farm and at POTW in NYS represent a significant opportunity for co-digestion of FSC resources (Chapter 1), yet little is known on the best ways to implement these mixtures (Chapter 4). Research into inhibition and synergistic effects should be conducted, including further development of the impact on hydrolysis rate. Another area of research to be considered is the development of small-scale anaerobic digestion. While the technology is fundamentally scalable current implementation trends favor large scale. A pairing of operational requirements (i.e. feedstock, digestate handling, etc.) to site specific needs (i.e., thermal demand vs. electricity, etc.) could result in increased penetration into smaller scale applications.
- **Environmental assessment of other utilization pathways**. Limited analysis exists regarding the environmental impacts of animal feed diversion (both wet and dry processes). The potential of diverting bakery FSC resources to animal feed (Chapter 5) warrants further development and research.
- **Development of the Organic Resource Climate Assessment Simulator (ORCAS) model.** The ORCAS model provides one of the first models in the US to estimate climate impacts across a variety of feedstock and a variety of pathways (Chapter 5). This model can be further developed to include additional pathways or integrated into other models to assess economic or other factors.

Appendices

Appendix A

Table A- 1: Vermont compost program FSC resources (g/student-day) [1]

	2013-14	2014-15	Average
student enrollment[1]	8097	7414	
lbs/school year[1]	249150	251,705	
lbs/student	30.78	33.9	
lbs/student-yr[2]	0.170	0.189	
g/student-yr	77.5	85.5	81.5

[1] Calculated from data obtained from Vermont compost program: file:///Users/jacquelineebner/Dropbox/Food%20Waste%20sources%20and%20stats/annual_compost_tonnage_report_2014.pdf
[2] Calculated assuming 180 day school year

Table A- 2: Florida K-12 food waste and milk waste study at elementary, middle and high school levels

Resource	Elementary (n=8) g/student-day		Middle School (n=7) g/student-day		High School (n=5) g/student-day	
Food	11.67	2.66	10.30	1.52	4.45	0.42
Milk	5.50	1.65	0.51	0.46	0.00	0.00

Source: Wilke et al., 2015

Table A- 3 Residential college and university meal audit data review (taken from Ebner et al., 2014)

Year	Institution	Pre-consumer food waste/meal (kg/meal)	Post consumer (plate waste)/meal (kg/meal)	Pre and Post Consumer Food Waste (kg/meal)	Liquid Waste/ meal (kg)[a]	Reference[c]
2004	North Michigan		0.13	0.17		Van Handel, B, 2004
2009-10	Universtity of Oregon	0.02	0.09	0.11		UO Campus Recycling Program, 2013
2010	Gettysburg College		0.07	0.09	0.04	Barresi M, et al.,, 2010
2011	Western Michigan			0.09	0.04	Merrow et al., 2012
2011	University of Virginia		0.06	0.07		Cochran, J., et al., 2011a, 2011b, 2011c.
2011	Colgate University		0.08	0.10	0.01	Burgett, et al., 2011
2011	Harvard		0.05	0.06		EPA website, 2011
2012	UC Davis	0.01	0.05	0.06	0.05	Jackson *et al, 2013*
2001	**CDEP formula**			**0.16**		**CDEP, 2001**
2012-13	Michigan State		0.10	0.13		Michigan State, 2013
2013	RIT			0.10		This Study
				0.10	**0.04**	**Mean**
				0.17	0.05	Max
				0.06	0.01	Min

[a] mass of liquid weight is calcualted assuming density of 1kg/L

[b] total pre and post consumer waste is calculated when not provided based upon the assumption that post consumer waste is 80% of total waste.

[c] For full references see Ebner, J., et al. "Estimating the biogas potential from colleges and universities." ASME 2014 8th International Conference on Energy Sustainability collocated with the ASME 2014 12th International Conference on Fuel Cell Science, Engineering and Technology. American Society of Mechanical Engineers, 2014.

Table A- 4: Meals per enrolled student data compiled through surveys and literature taken from Ebner et al., 2014[c]

Institutions	Meal transactions	Student enrollment	Meals per enrolled student (nearest 10)	year
Occidental	1079153	2100		
Cobbleskill	106505	2,470		
Middlebury	1612279	2516	640	2013
SUNY Cobleskill[ab]	272000	2600	100	2012
Sienna College[ab]	480000	3267	150	2012-13
St. John Fisher[a]	406400	4020	100	2013
UC Santa Cruz	2850134	16,753	170	2012-13
RIT	811,870	18,292	180	2013
UC Davis[ab]	1,800,000	31,426	60	2011
Purdue[a]	3,500,000	38,788	90	2013
Ohio University [ab]	3,800,000	56,387	70	2013
CDEP formula			**405**	
		mean	200	
		median	125	

[a] Dining service meals only

[b] Based upon weekly estimates, assuming 32 weeks/year

[c] see Ebner, J., et al. "Estimating the biogas potential from colleges and universities." *ASME 2014 8th International Conference on Energy Sustainability collocated with the ASME 2014 12th International Conference on Fuel Cell Science, Engineering and Technology*. American Society of Mechanical Engineers, 2014.

Table A- 5: Campus wide diversion statistics

Date	Institution	Food Waste (MT/yr)	Enrolled students+	Food Waste (kg)/enrolled student [a]	Source[c]
2009-10	Pamona College	26.36	1540	17.12	Miller, C. and Close, B., 2011
2012	Purdue	413.64	38788	10.66	Purdue University, 2013
2011	UC Davis	500.00	32354	15.45	UC Davis Sustainable Foodservice Progress Report, 2011
2012	Duke University	136.36	6655	20.49	Duke University, 2013
2012	Arizona State University (ASU)	1540.45	70000	22.01	Arizona State University, 2012
2011	Colgate University	74.55	2871	25.96	Burget et. al, 2011
2013	University of Washington	1155.71	43762	26.41	Newcomer, E, 2014
2013	RIT	499.22	18292	27.29	This study
2012	UC Santa Cruz	510.51	16753	30.47	UCDavis, 2014
2013	Dartmouth	239.09	6342	37.70	Dartmouth , 2013
2007	Stanford	1181.82	18136	65.16	Stanford, 2014
2012-13	Middlebury	326.77	2516	129.88	Biette, M. 2014

[c] see Ebner, J., et al. "Estimating the biogas potential from colleges and universities." *ASME 2014 8th International Conference on Energy Sustainability collocated with the ASME 2014 12th International Conference on Fuel Cell Science, Engineering and Technology*. American Society of Mechanical Engineers, 2014.

Table A- 6: Comparison of Connecticut (CDEP, 2001) and California (Calrecycle, 2014) studies on health and medical sector FSC resource generation factos

CDEP, 2001

	5.7 meals/bed-day *0.6lbs/meal *365 days/yr
Hospitals	5.7 meals/bed-day * 0. 272kg/meal * 365 days/yr
Nursing homes	3 meals/bed-day * 0.272 kg/meal* 365 days/yr

	lbs/bed-day	kg/bed-day	kg/bed-yr
Hospitals	5.7	0.27	**566**
Nursing homes	3	0.27	**298**

Calrecycle, 2014

	short tons waste/bed-yr	% food waste	short tons food waste/bed-yr	kg food waste/bed-yr
medical health sector	0.57	0.204	0.15	**137**

Used in this study			**140**

Table A- 7: Review of Hotel and Lodging resource generation data

	Category	Total waste per emp. yr (short tons)	% food waste	food waste per emp.-yr (short tons)	waste per guest room-yr (short tons)	food waste per guest room -yr	FSC resource generated (kg/emp-yr)
Calrecycle, 2014	hotel and lodging	2.14	28%	0.60	1.3	0.37	1945
Calrecycle, 2006	large hotels	2.52	39%	0.98	1.3	0.51	2295
Used in this study							**2100**

Table A- 8: Review of household FSC resource generation data

Composting program dataset (2014)

Sample Size	lbs/week	kg/week	kg/yr
200	11	4.989512	259

(Calrecycles, 2008)

	% food waste	tons per sector	households[a]	kg/household
Overall residential	25%	11,935,173	12,542,460	220

[a]Source: 2009-2013
households:http://quickfacts.census.gov/qfd/states/06000.html
2009-2013 Census

(Calrecycles, 1999)

	% food waste	tons per sector	households[a]	kg/household
Overall residential	20%	13,525,504	10,381,206	237

[a]US Census Bureau, Households and Families: Census Brief 2000;C2KBR/01-8, September 2001, https://www.census.gov/prod/2001pubs/c2kbr01-8.pdf

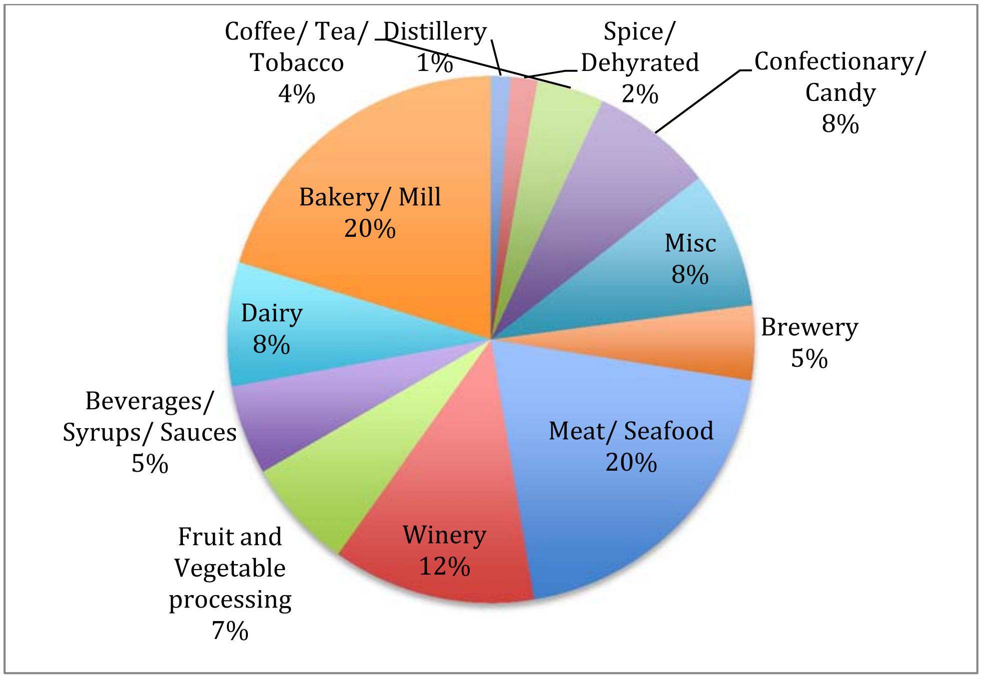


Figure A- 1: Distribution of NYS food manufacturers and processors

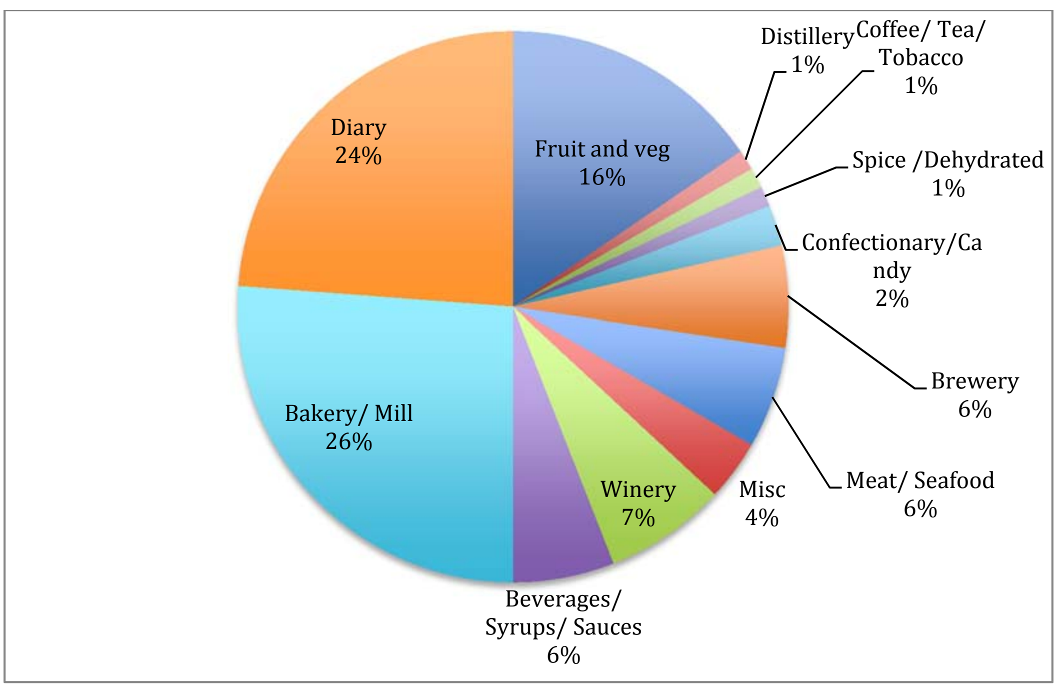


Figure A- 2: Distribution of data sample

Table A- 9: Summary of data collected on solid resources from food manufacturers and processors, including the number of establishments in the sample from each category, total solid resources reported (t/yr) from sample, description of the resources, utilization pathways reported, number reported generating solid resources, average quantity of solid resources (t/yr) generated by those establishments

Category	Sample	Reported solid resources (t/yr)	Description of resource	Reported treatments	Number reporting solid resources	Average resources (t/yr-establishment)
Fruit and veg processing	13	23,569	Scrap product or trimmings.	Ensiled or fed directly to animals (dry and wet), AD or land applied. Packaged product landfilled.	5	4,717
Distillery	1	0	None reported		0	-
Coffee/ Tea/ Tobacco	1	114	Chaf	Landfilled	1	114
Spice /Dehydrated	1	625	None reported			-
Confectionary/ Candy	2	88,220	None provided	Fed to animals	2	313
Brewery	5		Spent grain (treated and trapped as sludge).	Spent grain from brewing process to animals. Sludge from WWT, composted .	2	44,110
Meat/ Seafood	5	17	Bones, skins, fat, rejected product	Rendering company	1	17
Misc.	3	0	Not provided		0	
Winery	6	185	Skins, seeds, pomace	Composted, land applied also phenolic recovery from seeds	1	185
Beverages/ Syrups/ Sauces	5	29	Syrup, toppings	Dry animal feed	1	29
Bakery/ Mill	22	34,000	Stale, rejected porduct, crumbs.	Animal feed wet and dry.	15	2,267
Diary	20	56,286	Rejected product or WWT plant sludge	Fed to animals, land applied or landfilled if packaged	8	7,036
Total Sample	97	203,045			45	
Population	1092	776,603				

Table A- 10: Summary of data collected on low solid resources from food manufacturers and processors, including the number of establishments in the sample from each category, total solid resources reported (m3/yr) from sample, description of the resources, utilization pathways reported, number reported generating solid resources, average quantity of solid resources (m3/yr) generated by those establishments

Category	Sample	Reported low solid resources (m3/yr)	Description of resource	Reported treatments	Number reporting solid resources	Average resources (m^3/yr-establish ment)
Fruit and veg processing	13	1,839,516	Wastewater, liquid sludge	POTW and onsite WWT	8	229,940
Distillery	1	568	Spent grain	Fed to animals	1	568
Coffee/ Tea/ Tobacco	1	3,785	Wastewater, line change	POTW	1	3,785
Spice /Dehydrated	1	17,383	Process wash	Fed to animals	1	17,383
Confectionary/ Candy	2	-	none reported	POTW, onsite WWT(including AD) and land applied and land applied	0	-
Brewery	5	1,741,164	Wastewater, stillage	POTW and onsie WWT and land applie	4	435,291
Meat/ Seafood	5	113,967	Wash water	POW	4	28,492
Misc.	3	141,795	Not provided	POW	3	47,265
Winery	6	3,331	wash water, wine, lees	POTW	5	666
Beverages/ Syrups/ Sauces	5	167,288	Product and washwater	POTW, onsite WWT and land applied	4	41,822
Bakery/ Mill	22	416,291	Wastewater	POTW and land applied	10	41,629
Diary	20	1,418,218	Whey, wastewater, permeate, sludge	POTW, on-site WWT, land applied, fed to animals and off-site A	20	70,911
Total Sample	97	5,863,305			61	
Population	1092	22,425,871				

Table A- 11: Summary of data collect from NY State food banks and calculated statewide estimates

Food bank	Primary Prod. (t/yr)	Manuf. and Process. (t/yr)	Retail and Dist. (t/yr)	Other[a] (t/yr)	Total (t/yr)	Not Dist. (t/yr)	% t received
Long Island Cares, Inc.[b]	58	685	95	209	1,047	113	10%
Adjusted[a]	58	685	199	0	1,151		
% total	5%	60%	17%	0%	100%		
Food Bank of Western NY	638	959	1504	0	3,101	163	5%
Adjusted[a]	638	959	1504	0	3,101	163	5%
% total	21%	31%	48%	0%	100%		
Foodlink	193	525	495	3527	4,711	145	5%
Adjusted[a]	193	525	2259	0	6,475		
% total	3%	8%	35%	0%	100%		
Sum of reported	889	2,169	3,962	0	10,727		
Extrapolated	6,934	16,914	30,898	NE	83,656	4,905	6%
Regional Food Bank of Northeastern NY	NR	NR	NR	NR	13,818	909	7%
Sum of reported						1,330	6%

[a] In some cases an "Other" category was reported that included retail damage, food drives and walk-in donations and the "retail category" only included donated cases. In this case an adjustment was made to allocated 50% of the other category to retail as retail damage.

[b] Paper survey, 2013

[c] Online survey 2014

[d] 4 year average included data received for 2009-2012 via emal, hardcopy suvery in 2013 and online survey in 2014

[e] Grower's Harvest reported approximately 5,000t in 2015.

NE indicates not estimated

NR indicates not reported

The Golisano Institute for Sustainability and the New York State Pollution Prevention Institute at Rochester Institute of Technology are conducting research to help reduce organic pollutants and advance "green" industries. One area of focus is identifying alternative uses for organic wastes. In many cases these "wastes" can be valuable resources to other processes such as energy conversion. For this study, we are locating and characterizing organic wastes to find the best uses, considering costs, energy use and overall environmental impact.

To participate, please complete the information requested below and return this form to: nysp2i@rit.edu. We appreciate your support of our effort to assist New York State's food processing industry – a vital component of our economy and community. For additional information contact Jacqueline Ebner (jhe5003@rit.edu, 585-899-0151).

Company Information

Company Name:

Address 1 :

Address 2 :

City / Town :

County : Zip code :

Contact Name:

Email Address:

Phone Number:

Please describe the company's activities/main products:

Organic Waste (/Organic Resources)

Please provide as much information as possible regarding the organic (i.e. biodegradable) waste/resources generated at your facility. This includes organic waste that is recycled, donated, composted, disposed, etc. i.e. all the waste that is produced regardless of how it is disposed or where it ends up. DO NOT include waste such as cardboard, office paper (except food-soiled paper), textiles, plastic, metals, etc. If packaged or mixed with inorganics please indicate so and estimate the percentage of the waste that is organic.

Organic resources can have high solid content (>25% solids) or low solid content (<25% solids). Examples include: wastewater, oil/grease, trimmings, production waste/by-products, food-soiled paper, and residue (leaves, stems).

High Solid content organic Waste (tons/year) or indicate units	**% Organic (if packaged or mixed with non organics)**	**Organic Waste Description (ie: orange peels, canned out-of-spec vegetables, unpackaged rejected desert topping, pomace, bottled soda, spent grains,etc.)**	**Current waste treatment (compost, land application, landfill, animal feed, donation, etc)**	**Indicate Estimate (E) or Actual (A)**	**Disposal charge paid ($/year)**
Low solid Organic Wastewater (gal/year)	**Average BOD (mg/L)**	**Organic Wastewater Description (ie: whey, process wash from fruit canning line, beer stillage, oil, grease trap waste, etc.)**	**Waste treatment (discharge to sewer, onsite treatment, land application, septic, etc.)**	**Indicate Estimate (E) or Actual (A)**	**Surcharge or fees ($/year)**

The information you provide in this document will be used to estimate regional biomass resource potential. Any information made available to the public will be aggregated; data will NOT be disclosed at the individual company level without company consent.
Request for Company Consent: Please indicate below whether you agree to allow the information you have provided in this survey to be shared withcompanies/organizations interested in utilizing organic waste.

☐ *Yes - I consent to the distribution of my company's data provided in this survey*

☐ *No – I do not wish to disclose my company's data provided in this survey with outside companies/organizations.*

Name / title ______________________________ . _*Date*__________

We would like to collect a sample of your organic waste for characterization, which will help determine optimal alternative uses. Please provide a contact we may call to get a sample for characterization.

Name: ______________________________ *Phone:* ________________ *Email:*________________________

Appendix B

Table B- 1: Literature review of lifecycle assessment studies of anaerobic digestion

Study[a]	Region	Type of study	Feedstock	System	Scope/FU/ system boundary
Artrip et al., 2013	U.S., Pacific NW	Case study LCA	Manure monodigestion	Plug flow system	Tier 1 and 2 per IPCC, no feedstock,no digesate storage or LA
Bacenetti et al. 2013	Northern Italy	Case study for CED and GWP of electricity production for 3 plants in Italy	Maize silage mono, pig slurry mono and co-digestion.	250, 520 and 999 kW, CSTR w solid separation and recycling of liquid effluent for dilution.	Cradle to grave, FU=1kWe, from crop cultivation and slurry collection to digestate management, ref systems considered.
Bartram and Barbour, 2004	U.S., Chino Basin, CA	Analysis to estimate GHG reduction due to system implementation	Dairy manure	Centralized complex of digesters in chino basin	Includes FW tranport but no ref case. Ref case for manure includes storage, includes enteric ferm and coral emissions. No digestate storage or land application
Bentley et al., 2010	U.S., NY	Scenario analysis of different system implementation	Manure and co-digestion of organic waste	Community scale digester, electricity to grid	Annual emissions of GHG. Doe not include include ref case food waste treatment does include ref case manure mgt., no land application or fert displacement
Boldrin et al., 2011	Denmark	Decision support tool modeling various waste treatments	MSW, allows for specification of characteristics and includes broad category of food waste	Various options	Model with varying inputs related to feedstock composition, can include MRF module and options for managment of digestate

Borjesson and Berglund, 2006	Swedish conditions, SOA technology	Comparitive LCI based upon literature	mono-substrate; MOW, IFW, manure, harvest residues, ley crops,	small scale and large scale, upgraded for transport, heat and power and heat.	Cradle to gate emissions (LCI); FU=MJ energy service provided;upstream impacts of cultivation and harvest for energy and ley crops but only collection for MSW and transport for FIW. No reference system for manure, crops or wastes.
Borjesson and Burglund, 2007	Swedish conditions, SOA technology	Comparitive LCA based upon literature	same as above	same as above	Cradle to grave for several impact categories (LCA). upstream impacts of cultivation and harvest for energy and ley crops as well as fertilizer recycing and use. For MSW assumes ref case of combustion/ composting of ash and composting IFW and fossil fuel for energy equivalent.
de Vries and de Boer, 2010	Western Europe	Data gathered from literature and expertise			Comparison of impacts due to digestate (FU- 1 ton of applied product) several environmental impacts.)
Ishikawa et al., 2006	Japan	Data from centralized plant			Not clear
Dressler et al., 2012	Germany	Comparative LCA of influence of regional parameters soil,climate, fuel use and irrigation based on data from other studies for regions	maize	3 AD systems from 3 regions of Germany	Cradle to grave (silage considered a waste, no burden and no avoided burden), 2 FUs, 1 kg maize and 1 kWhe, assumes airtight storage

Edelmann et al., 2004	Switzerland	Comparison of pathways to treat household organic waste, data from previous studies (Edelmann, 2000) soil prop, xport dist.	OFMSW w pig and dairy manure	Described in 2000 paper (na)	EcoIndicator single score FU=10,000 tons household waste
Fuchsz and Kohlheb, 2014	Germany	Comparative study comparing feedstocks based upon European datasets	Energy crop, crop manure and slurry and slury and manure	Modeled wet 2 staged CSTRs	cradle to gate, FU=1kWe, plant construction,feedstock production, biogas production and electricity production. Land application considered equivalent to ref case
Jury et al., 2010	Luxembourg	Case study based on pilot and lab scale fermentation w field scale crop cultivation for biogas injection compared to natrual gas	Energy crops	Injection to the grid	FU=1MJ injected natural gas.
Lijo et al., 2014a	Italy	Case study LCA 2 AD plants	Mono- maize, pig slurry.	500kW, 250kW	call it "cradle to gate" but really cradle to grave?: FU= 1t feedstock mixture, 6 impact categories
Lijo et al., 2014b	Italy	Case study LCA	Co-digestion pig slurry and energy crops	100kWe system	call it "cradle to gate" but really cradle to grave?: Includes crop cultivation and recycling to fields
Moller et al., 2009	Europe	Comparative study of impacts of digestate use based upon literature		source separated MSW	A variety of AD plants and end uses. FU- 1 ton of applied product

Poeschl et al., 2012a	German, SOA plant	Comparitive LCI literature/ecoInvnet	mono and co-digestion w/ manure, several energy crops and several FW (MSW, pomace, GTW, slaughter house waste)	small(<500kWe) and large (>500kWe) scale, fuel cell Stirling engine, and micro gas turbine also variety of digestate treatment and handling	FU=1t Feedstock digested. System expanded for fertilizer displacement and electricity generated but not waste treatment.
Poeschl et al., 2012b	German, SOA plant	Comparative LCA literture/ecoInvent		same as above	FU=1t Feedstock digested. System expanded for fertilizer displacement and electricity generated but not waste treatment.
Pronto and Gooch, 2010	NYS	data from 7 farms	manure co-dig w/ organic waste	boiler and 200kW genset and flare	Gate to gate, based on CAR, RGGI, CCX and EPA protocols, used data on feedstock but not on digestate
Rodriguez-Verde et al., 2014	Northwest Spain	Comparative/feasability study of AcoD w agro wastes	Pig manure PM reference case; PM with MW (molasses waste) and FW(fish waste); co-digestion of PMwith BW(biodiesel waste); co-digestion of PM with VW (vinasse waste).	Cenralized, (CSTR) w energy for pumping and pasturization and converting biogas to electricity	FU=110,000 t/yer of PM,regional xport parameters and PM characteristics, lab B0
Wulf et al., 2006	Germany	Comparative study compiled from literature to identify measures to reduce GHG of AcoD	Pig slurry and OFMSW	A model mesophyllic plant	Gate to grave, Fermentation, Storage and Land application and fertilizer displacement only considered. (Not, avoided disposal or xport to the facility or storage (assumed airtight)).

Nomenclature

$B_{0.M}$= Bio-methane potential of manure (m_3CH_4/t)
$CF_{C\text{-}CO2}$= Conversion factor for carbon to CO_2e (fraction)
$CF_{N2O\text{-}N}$= Conversion factor for N_2O –N to N_2O (fraction)
$CH4_{LFG,i}$= Methane recovered by LFG system for substrate i (m^3)
CS_{LA}=Portion of carbon sequestered after land application (kgC/kg VS)
CS_{LF}=Portion of carbon sequestration from landfilling (kgC/kg VS)
$CH4_{LFG,i}$= methane generated from landfill of annual mass of waste i(m^3/yr)
D_{Feed}= displacement of animal feed (kg maize feed/kg dry dairy waste)
CE= Electricity conversion efficiency factor (fraction)
EC=Energy content of CH_4 (BTU/m3 CH4)
EF_1=Portion of N stored emitted as direct N_2O-N during land application (kg N_2O-N/ kg N)
EF_3=Portion of N stored emitted as direct N_2O-N (kg N_2O-N/kg N)
EF_4=Portion of N volatilized emitted as N_2O-N (kg N_2O-N/ kg N)
EF_5=Portion of N leach/runoff emitted as N_2O-N (kgN_2O-N/kg N)
EF_{feed}= GHG emissions due to cultivation and production of maize animal feed (kg CO_2e/kg feed)
$EF_{freight}$ = Fuel lifecycle GHG emissions for a combination truck, short haul, diesel powered Northeast (kgCO_2e/t*km)
$EF_{grid,NPCC}$=Emission factor for regional grid emissions (kg CO_2e/MWh)
$EF_{N2O,IF}$= Emission factor for indirect and direct emissions due to inorganic fertilizer application (kgCO2e/kg Neff)
EF_N=Emission factor for synthetic Nitrogen fertilizer production (kgCO_2e/kg N)
EF_{OP}=Emission factor for fossil fuel due to transport of waste to the landfill and operation of the landfill (kg CO2e/twaste)
EF_P=Emission factor for synthetic P fertilizer production
EF_{spread}=Emission factor for transport and application of organic fertilizer to land (kg CO_2e/t)
EF_{WWT}=Emission factor for disposal of wastewater at a municipal WWTP (tCO_2e/t)
EM_{AcoD}= GHG emissions due to Anaerobic Co-digestion case (tCO_2e/yr)
$EM_{CS,LA,M}$=GHG impact due to sequestered carbon from land application of manure (tCO_2e/yr)
$EM_{CS,LA,D}$=GHG impact due to sequestered carbon from land application of digestate (tCO_2e/yr)
$EM_{CS,LA,dairy}$=GHG impact due to sequestered carbon from land application of dairy waste (tCO_2e/yr)
$EM_{CH4,i}$= GHG impact due to uncaptured methane emissions from landfill of waste (i) (tCO_2e/yr)
$EM_{fert,M}$=GHG emissions due to displacement of inorganic fertilizer from manure land application (tCO_2e/yr)
$EM_{fert,D}$=GHG emissions due to displacement of inorganic fertilizer from digestate land application (tCO_2e/yr)
$EM_{fert,dairy}$=GHG emissions due to displacement of inorganic fertilizer from dairy waste land application (tCO_2e/yr)
EM_{FD}=GHG emissions due to food waste disposal (t CO_2e/yr)
$EM_{grid,NPCC}$= Displaced non-baseload emissions for NY State regional grid mix (NPCC) (tCO_2e/yr)
$EM_{LA,D}$= GHG emissions due to land application of digestate (kg CO_2e/yr)
$EM_{LA,dairy}$= GHG emissions due to land application of dairy waste (kg CO_2e/yr)
$EM_{LA,M}$= GHG emissions due to land application of manure (tCO_2e/yr)
$EM_{LA,N2O,D}$= GHG emissions due to direct and indirect N_2O from land application of digestate (tCO_2e/yr)
$EM_{LA,N2O,dairy}$= Emissions due to direct and indirect N_2O from land application of dairy waste (tCO_2e/yr)
$EM_{LA,N2O,M}$= GHG emissions due to direct and indirect N_2O from land application of manure (tCO_2e/yr)
$EM_{LA,N2Od,D}$=Impact of direct N_2O emissions due to land application, digestate (tCO_2e/yr)
$EM_{LA,N2Od,dairy}$=Impact of direct N_2O emissions due to land application, dairy waste (tCO_2e/yr)
$EM_{LA,N2Od,M}$=Impact of direct N_2O emissions due to land application, manure (tCO_2e/yr)
$EM_{LA,vol,,D}$=Impact of indirect N_2O emissions due to volatilization at land application, digestate (tCO_2e/yr)
$EM_{LA,vol,dairy}$=Impact of indirect N_2O emissions due to volatilization at land application, dairy waste (tCO_2e/yr)
$EM_{LA,vol,M}$=Impact of indirect N_2O emissions due to volatilization at land application, manure (tCO_2e/yr)
$EM_{LA,runoffleach,D}$=Indirect N_2O emissions due to leaching/runoff, digestate (tCO_2e/yr)
$EM_{LA,runoffleach,dairy}$=Indirect N_2O emissions due to leaching/runoff, dairy waste (tCO_2e/yr)
$EM_{LA,runoffleach,M}$=Indirect N_2O emissions due to leaching/runoff, manure (tCO_2e/yr)
EM_{LF},i= Emissions due to landfilling of food waste (i) (tCO^2e/yr)
$EM_{LFG,i}$= Emissions avoided due to electricity generated through LFG from disposal of waste (i) (tCO_2e/yr)
EM_{OP}= Emissions due to transport of waste to landfill and operation of the landfill
EM_{RC} = Reference Case GHG emissions (tCO_2e/yr)
$EM_{ST,D}$ =GHG due to emissions due to storage of manure (t CO_2e/yr)
$EM_{ST,M}$ =GHG due to emissions due to storage of manure (t CO_2e/yr)
$EM_{ST,CH4,D}$=GHG emissions due to CH_4 during storage of manure(tCO_2e/yr)
$EM_{ST,CH4,M}$=GHG emissions due to CH_4 during storage of manure(tCO_2e/yr)
$EM_{ST,N2O,D}$ =GHG due to emissions due to storage N_2O emissions manure (tCO_2e/yr)
$EM_{ST,N2O,M}$ =GHG due to emissions due to storage N_2O emissions manure (tCO_2e/yr)
$EM_{ST,N2Od.,D}$ =GHG emissions due to direct N_2O emissions during storage of manure (tCO_2e/yr)
$EM_{ST,N2Od.,M}$ =GHG emissions due to direct N_2O emissions during storage of manure (tCO_2e/yr)
$EM_{ST,runoffleach,D}$= GHG emissions due to indirect N_2O from runoff
and leaching during storage of manure (tCO_2e/yr)
$EM_{ST,runoffleach,D}$= GHG emissions due to indirect N_2O from runoff
and leaching during storage of manure (tCO_2e/yr)

$EM_{ST,runoffleach,M}$= GHG emissions due to indirect N_2O from runoff
and leaching during storage of manure (tCO_2e/yr)
$EM_{ST,vol,M}$ = GHG emissions due to indirect N_2O emissions from volatilization of N during storage of manure (tCO_2e/yr)
$EM_{spread,D}$= GHG impact of applying digestate to land (tCO2e/yr)
$EM_{spread,dairy}$= GHG impact of applying dairy to land (tCO2e/yr)
$EM_{spread,M}$= GHG impact of applying manure to land (tCO2e/yr)
$EM_{spread,D}$= GHG impact of applying manure to land (tCO2e/yr)
$EM_{spread,M}$= GHG impact of applying manure to land (tCO2e/yr)
$EM_{WWT/LF,\ i}$= Emissions due to disposal of FOG($kgCO^2e$/yr)
$EM_{WWT,\ i}$= Emissions due to disposal at WWTP) ($kgCO^2e$/yr)
$EM_{xport,AF}$=Emissions due to transport of dairy waste to feed animals (tCO_2e/yr)
$EM_{xport,LA}$=Emissions due to transport of dairy to fields (tCO2e/yr)
$EM_{xport,WWTP}$=Emissions due to transport of FOG to WWTP (tCO2e/yr)
$Elec_{LFG,I}$ = electricity produced via LFG recovery for waste i(kWh)
$Frac_{GASM}$= Portion of N that is volatilized as NH_3 during land application of manure (fraction)
$Frac_{GASM.;s}$= Portion of N that is volatilized as NH_3 during storage for a liquid slurry
of manure (fraction)
$Frac_{GASwhey}$=Portion of N that is volatilized as NH3 during land application of whey
$Frac_{runoff,ls,ma}$ = Portion of N that is lost as runoff, for liquid slurry in the mid atlantic region (fraction)
GC_{NY}=portion of landills with LFG recovery for NY (fraction)
GWP_{CH4}=Global Warming Potential of CH_4
GWP_{N2O}=Global Warming Potential of N_2O
HR= heat rate of LFG to energy conversion (BTU/kKWh)
K_{FW}= decay rate constant for food waste
L_{0i}=methane production potential (m^3CH_4/t)
LCE= Landfill gas capture efficiency (fraction of gas captured each year)
MinF,$_{dairy}$= Mineralization factor (%N)
t_{IN}=Annual influent biomass (t)
t_M= Annual mass of manure (t)
t_D=Annual mass of digestate (t)
t_{dairy}= Annual mass dairy wastewater influent
$MCF_{ls,NY}$= Estimated Methane Conversion factor for liquid slurry storage in NY
$Minfactor_{NY,M}$= Mineralization of organic N after 3 years, NY (fraction)
$N_2O\text{-}N_{ST,D}$= N lost as N_2O during storage,digestate (kg N/yr)
$N_2O\text{-}N_{ST,M}$= N lost as N_2O during storage, manure (kg N/yr)
N_{2loss}= Portion of N_2 (fraction)
$N_{LA,D}$= N land applie, digestate (net of storage N losses) (kg N/yr)
$N_{LA,M}$= N land applied, manure (net of storage N losses) (kg N/yr)
$N_{LA,runoff,D}$=N loss due to runoff during land application of digestate (kg N/yr)
$N_{LA,runoff,M}$=N loss due to runoff during storage of manure (kg N/yr)
$NMin_{,LA,M}$=Mineral N land applied (kg N/yr)
NO_{loss}= Portion of N lost as NO (fraction)
$N_{ST,runoff,M}$= N loss due to runoff during storage of manure (kg N/yr)
$N_{STrunoff,D}$=N runoff during digestate storage (kg N/yr)
$N_{ST,vol,D}$= N volatilized during storage of digestate (kg N/yr)
$N_{ST,vol,M}$= N volatilized during storage of manure(kg N/yr)
$N_{org,LA,M}$= Organic N land applied (kgN/yr)
N_{effM}= Effective inorganic fertilizer displaced (kgN/yr)
OX= fraction oxidized (fraction)
P_{eff}= Plant available portion of applied P (fraction)
RHO_{CH4}=density of CH_4 (kg CH_4/m^3 CH_4)
TKN_M= concentration of N, manure (mg N/kg)
TKN_D= concentration of N, digestate (mg/kg)
TKN_{dairy}= concentration of N, dairy wastewater (mg/kg)
VS_M = Volatile solids content of manure (gVS/kg)
VS_D=Volatile solids content of digestate (gVS/kg

Table B- 2: GHG model formulae, methodology and parameters, sources and uncertainty

Emission Source	Emissions formulae (t CO_2e/yr) and reference (source)	Parameters (Table 3-1 unless noted)	Emission factors (source)	Uncertainty Analyzed
Reference Case (EM_{RC})	$EM_{RC} = EM_{ST,M} + EM_{LA,M} + EM_{FD}$		$GWP_{CH4}=28$[15] $GWP_{N2O}=265$[15] $CF_{N2O\text{-}N}= 44/28$ $RHO_{CH4}=0.67 kgCH_4/m^3CH_4$ $CF_{C\text{-}CO2}= 44/12$	
Manure Storage	$EM_{ST,M} = EM_{ST,CH4,M} + EM_{ST,N2O,M}$[17]			
Storage CH_4, manure	$EM_{ST,CH4,M}=B_{o,M}*RHO_{CH4}*VS_M*t_M*MCF_{ls,NY}*GWP_{CH4}$	$B_{o,M}$[24], VS_M, t_M	$MCF_{ls,NY} =0.24$[18 (Table A-203)]	MCF=+-20%[10]
Storage N_2O manure	$EM_{ST,N2O,M}= EM_{ST,N2Od,M}+ EM_{ST,vol,M} + EM_{ST,runoff,ls,ma}$			
	$EM_{ST,N2Od,M} =N_2O\text{-}N_{ST,M} * CF_{N2O\text{-}N} *GWP_{N2O}$ $N_2O\text{-}N_{ST,M}= EF_3*TKN_M*t_M$[17]	TKN_M, t_M	$EF_3=0.005$[17,18 (Table A-204)]	EF_3=Factor of 2[17]
	$EM_{ST,vol,M}= N_{ST,vol,M} *EF_4* CF_{N2O\text{-}N}*GWP_{N2O}$ $N_{ST,vol,M}= Frac_{GASM.ls} * TKN_M*t_M$	TKN_M, t_M	$EF_4=0.01$[17,18] $Frac_{GASM,ls}=0.26$[18(Table A-205)]	EF_4=0.002 - 0.050[17,18] $Frac_{GASMS}$= 0.15-0.45[17,18]
	$EM_{ST,runoffleach,M} =N_{ST,runoff,M}* EF_5*CF_{N2O\text{-}N}*GWP_{N2O}$ $N_{ST,runoff,M}=Frac_{runoff,ls,MA}*TKN_M*t_M$	TKN_M, t_M	$EF_5=0.0075$[17,18] $Frac_{runoff,ls,MA}= 0.007$[18]	$Frac_{runoffleach}$=0-0.3[18]
Land Application-manure	$EM_{LA,M}=EM_{spread}+EM_{LA,N2O,M}-EM_{fert,M}+EM_{CS,LA,M}$			
Spreading of manure	$EM_{spread,M}=EF_{spread} *t_M$	t_M	EF_{spread}=0.8kg	$EF_{spread,M}$=0.9-1.9[12]
N_2O land application	$EM_{LA,N2O,M}=EM_{LA,N2Od,M}+EM_{LA,vol,M}+EM_{LA,runoffleach,M}$[17]			

	$N_{LA,M}$= TKN_M*t_M -N_2O-$N_{ST,M}$- $N_{STvol,M}$- $N_{ST,runoff,M}$-(NO_{loss}+N_{2loss})* TKN_M*t_M $NMin_{,LA,M}$=(TAN_M*t_M)-N_2O-$N_{ST,M}$- $N_{STvol,M}$-$N_{ST,runoff,M}$-(NO_{loss}+N_{2loss})* TKN_M*t_M $N_{org,LA,M}$=(TKN_M-TAN_M)* t_M	TKN_M, t_M, TAN_M	NO_{loss} =0.012[20] N_{2loss}= 0.008[19]	
	$EM_{LA,N2Od,M}$= N_2O-$N_{LA,M}$* $CF_{N2O-N-N}$* GWP_{N2O}[17] N_2O-$N_{LA,M}$= EF_1*$N_{LA,M}$[10]		EF_1=0.0125[17,18 (Table 11.1)]	EF_1= 0.005-0.05[17]
	$EM_{LA,vol,M}$ = $N_{LA,vol,M}$* EF_4*CF_{N2O-N}* GWP_{N2O} $N_{LA,vol,M}$= $N_{LA,M}$ * $Frac_{GASM}$		EF_4=0.01 [17,18(Table 11.3)] $Frac_{GASM}$=0.20[17,18 (Table 11.3)]	EF_4=0.002 - 0.050[17] $Frac_{GASM}$= 0.05-0.50[17]
	$EM_{LA,runoffleach,M}$ = $N_{LA,runoff,M,}$ * EF_5* CF_{N2O-N} * GWP_{N2O} $N_{LA,LEACH,M,}$= $Frac_{runoff,ls,MA}$ *$N_{LA,M}$		EF_5=0.0075[17 (Table 11.3)] $Frac_{runoff,ls,MA}$= 0.007[17]	$Frac_{LEACH}$=0-0.8[17] EF_5=0.005-0.025[17]
Fertilizer displacement, manure land application	$EM_{fert,M}$=$Neff_M$*(EF_N+$EF_{N2O,IF}$)+$Pmin_{factor}$*P_M*t_M* EF_P N_{effM}= $N_{org,LA}$*$Minfactor_{,NY}$+ $Nmin_{LA,M}$–N_2O-$N_{LA,M}$-$N_{vol,LA,M}$-$N_{LA,runoff,M}$	TKN_M, t_M, TAN_M, P_M	$Min_{factor,NY,M}$=0.52[21] $Pmin_{factor}$=0.90[22] EF_N=6.8[17] $EF_{N2O,IF}$=5.[23] EF_P=0.41[22]	
Carbon Sequestration, manure land application	$EM_{CS,LA,M}$=CS_{LA}*VS_M*t_M *CF_{C-CO2}	VS_M,t_M	CS_{LA}=0.13[22,24]	CS=+/- 20%
Food Disposal –	$EM_{FD} = \sum EM_{LA,i} + EM_{LF,i} + EM_{WWT/LF,i} + EM_{WWTP,i}$ where i indicates food waste category			
Land application-	$EM_{LA,dairy}$=$EM_{xportLA,i}$+$EM_{LA,i}$			
Transport of food waste to the fields ($EM_{xport,dairy}$)	$EM_{xport,LA,i}$=$EM_{freight}$*t_{diary} *km_{LA}	t_{dairy}[a] km_{LA}=100 (assumed)	$EF_{freight}$=0.107 [26]	

Land Application,dairy	$EM_{LA,diary}=EM_{spread,dairy}+EM_{LA,N2O,dairy}-EM_{fert,diairy}+EM_{CS,LA,dairy}$			
	$EM_{spread,dairy}=EF_{spread}*t_{LA,dairy}$	t_{dairy}[a]	$EF_{spread}= 1.5$[22 d]	
Land application N_20	$EM_{LA,N2O,dairy}=EM_{LA,N2Od,dairy}+EM_{LA,vol,dairy}+EM_{LA,runoffleach,dairy}$[10]			
	$EM_{LA,N2O,,dairy}=EF_{1,}*TKN_{dairy}*t_{dairy}+CF_{N2O-N}*GWP_{CH4}$	TKN_{dairy}[b] t_{dairy}[a]	$EF_1=0.0125$[17,18(Table 11.1)]	
	$EM_{LA,vol,dairy}=EF_4*TKN_{dairy}*t_{dairy}*Frac_{GASwhey}*CF_{N2O-N}*GWP_{N2O}$	TKN_{dairy}[b] t_{dairy}[a]	$Frac_{GAS,dairy}= 0.006$[25]	
	$EM_{LA,runoffleach,dairy}=EF_5*TKN_{dairy}*t_{dairy}*Frac_{LEACH-whey}*CF_{N2O-N}*GWP_{N2O}$	TKN_{dairy}[b] t_{dairy}[a]	$Frac_{runoffdairy,}= 0.008$[25]	
Fertilizer displacement	$EM_{fert}=t_{diary}*MinF_{dairy}*TKN_{diary*}(EF_N+EF_{N2OIF})$ $P_{dairy}*t_{dairy}*EF_P$	TKN_{dairy}[b] t_{dairy}[a] P_{dairy}[b]	$Min_{F,\ dairy} = 0.20$[25] $P_{eff}=0.90$[25] $EF_N=6.8$[17] $EF_{N2OIF}=5.4$[23] $EF_P=0.41$[23]	
Carbon Sequestration,diary land application	$EM_{CS,LA,dairy}=CS_{LA}*VS_{dairy}*t_{dairy*}CF_{C-CO2}$	VS_{dairy}[b], t_{dairy}[a]	$CS_{LAdairy}=0.10$[24]	
Food disposal GTW/DAF-	$EM_{WWTP/LF}=EM_{xport,,WWTP}+EM_{LF,i}$			
	$EM_{xport,,WWTP}=EF_{freight}*t_{GTW/DAF}*km_{,WWTP}$	$t_{GTW/DAF}$[a] $km_{,WWTP}=50$ (assumed)	$EF_{freight}=0.107$[26]	
Emissions Landfill – $EM_{LF,i}$ where i= waste type	$EM_{LF,i}= EM_{OP}+ EM_{CH4,i}-EM_{LFG,i}- EM_{CS,LF,i}$ $EM_{OP}=EF_{OP*}t_{,i}$	$t_{GTW,}$[a] $t_{DAF,}$[a] $t_{FW,}$[a]	$EF_{OP}=44$[30]	

Uncaptured methane emissions	$EM_{CH4,i} = L_{0,i} * t_i * RHO_{CH4} * \sum_{x=1}^{30}[(e^{-k(x-1)} * (1-e^{-k}) * (1-OX) * (1-(GC_{NY} * LCE)]$ $*GWP_{CH4}$[31,32]	$L_{0,GTW}$[bc] $L_{0,DAF}$[bc] $L_{0,FW}$[bc]	$K_{foodwaste}=0.144$ [3025] $GC_{NY}=0.9$[30,31] LCE=Year 1=0, year 2=45%, year 3=60%, year 4=65%, year5=70%, year 6-11=75%, year 12=79%, year 13=83%, year 14=87%, year 15=91%, year 16-30=95% [40] OX=0.1 [294]	
Avoided emissions due to LFG recovery	$EM_{LFG,i} = Elec_{LFG,i} * EFgrid$ $Elec_{LFG,i}=L_{0,i}*ti*\sum_{x=1}^{30}[(e^{-k(x-1)}) *(1-e^{-k}) *(1-OX)*(GC*LCE)]* CF *EC/CE$		EC=35315 BTU/$m^3 CH_4$[30] CF =0.85 [30] CE=11700 BTU/KWh[30] EF_{grid}=-0.534[41]	
Carbon Sequestration	$EM_{CS,LF}=CS_{LF}* VS_i*t_i$	VS_i[bc], t_i[ac]$L_{0,GTW}$[c]	CSfoodwaste=0.08kg/kg dry waste[42]	
WWT emissions	$EM_{WWT}=EF_{WWT}*t_{WWT}/1000EM_{LFG,i}$	t^a_{WWT}	$EF_{WWT}=0.518$[43]	
Animal Feed Displacement	$EM_{AF,dairy}=EM_{xport,AF}- (D_{Feed}*TS_{dairy}*t_{dairy}* EF_{Feed})$	t_{diary}[a] km_{dairy}= 100km (assumed) TS_{dairy}=0.04[b25]	$EF_{freight}$=0.107[26] D_{Feed}= 1.27[44] EF_{feed}=0.592[43]	
AcoD Case	$EM_{AcoD}=EM_{xport,FW}+EM_{DIG}+EM_{ST, D}+EM_{LA,D}EM_{WWT}$			
FW hauling	$EM_{xport,,FW}=EF_{freight} *(t_i*km_i)$	t_i[ab] , $Km,_i$[cb]	$EF_{freight}$=0.107/t food waste[26a]	
Digester Emissions AcoD Case	$EM_{DIG}=EM_L+EM_{IC}-EM_{GRID}$ $EM_{AcoD}=EM_{xport,FW}+EM_{DIG}+EM_{ST, D}+EM_{LA,D}$			

Fugitive Emissions (EM_{leak})FW hauling	$EM_{Leak}= EF_{leak}*Q_{CH4}* CF_{CH4}*GWP_{CH4}$ $EM_{xport,,FW}=EF_{freight} *(t_i*km_i)$ where i=food waste deliveries	$Q_{CH4}t_i$[b] , $Km,_i$[b]	EF_{leak} =3% methane utilized[15],[46] $EF_{freight}$=0.107/t food waste[2218]	EF_{leak} =0-10%[15]
Disgester Emissions	$EM_{DIG} =EM_{Leak}+EM_{IC}-EM_{grid}$		EFLeak=3%45,46	
Fugitive Emissions (EM_{leak})	$EM_{Leak}= EF_{Leak}*Q_{CH4}* CF_{CH4}*GWP_{CH4}$	Q_{CH4}	EF_{Leak}=3%[45,46]	F_{leak} =0-10%[17]
Incomplete combustion (EM_{IC})	$EM_{IC}=EF_{IC,CH4}*Q_{CH4}*RHO_{CH4}*GWP_{CH4}+ EF_{IC, N2O}*Q_{CH4}/1000* GWP_{N2O}$	$EF_{IC,CH4}$ =2.5% [d](this study) $EF_{IC, N2O}$ =0.03[d] g N_2O/m^3 CH_4 Q_{CH4}	$EF_{IC,CH4}$ =2.5% methane utilized [d] (this study) $EF_{IC, N2O}$ =0.03[d] $gN_2O/m3$ CH_4	$EF_{IC,CH4}$ =0.4%-3.28%[43,45] $EF_{IC, N2O}$ =0.02-1.75g [43,45]
Displaced grid emissions (EM_{grid})	$EM_{grid}=(MWh_{grid}- MWh_{parasitc})*EF_{grid}$	MWh_{grid}	$EF_{grid,NPCC}$= (533.66)[41]	$EF_{grid,National average}$= -(689.53) $kWCO_2e/MWh$[41]
Digestate Storage	$EM_{ST,D} = EM_{STCH4,D} + EM_{STN2O,D}$[10]			
Digestate storage CH_4	$EM_{ST,CH4,D}= EF_{CH4,D}*VS_{,D}* t_D* RHO_{CH4}* GWP_{CH4}$[10]	VS_D, t_D	$EF_{CH4,D}$ =0.54 this study[51-54]	$EF_{CH4,D}$=0.004 -0.074 m^3CH_4/kg VS[48-51]
Digestate Storage N_2O	$EM_{ST,N2O,D}= EM_{N2Odirect,ST,D}+ EM_{N2Ovol,ST,D}+ EM_{N2OSTrunoffleach,D}$[10]			

	$EM_{N2O,ST,D}$ $=N_20-N_{ST,D}$ * CF_{N2O-N} $*GWP_{n2o}$ $N_2O-N_{ST,D}$= $EF_3*TKN_D*t,_D$	TKN_D, t_DVS_D, t_D	EF_3=0.005[17,18]	EF_3=Factor of 2[15] EF_1= 0.05-0.5[17]
	$EM_{ST,vol,D}$= $N_{vol,ST,D}$ $*EF_4*$ $CF_{N2O-N}*GWP_{N2O}$ $N_{vol,ST,D}$= $Frac_{GASMS}$ * $TKN_D*t,_D$ $EM_{ST,N2O,D}$= $EM_{N2Odirect,ST,D}$+ $EM_{N2Ovol,ST,D}$+ $EM_{N2OSTrunoffleach,D}$[10]	TKN_D, t_D	EF_4 =0.01[17] $Frac_{GASMS}$=0.26[17,18] (Table A-205)	EF_4=0.002 - 0.050[17] $Frac_{GASM}$= 0.05-0.50[17]
	$EM_{N2OSTleachrunoff}$=$N_{leachrunoff}$* $EF_5*CF_{N2O-N}*GWP_{N2O}$ $N_{runoffleach}$=$Frac_{runoffleachST,M}*TKN_D*t,_{D/}1000EM_{N2O,ST,D}$ $=N_20-N_{ST,D}$ * CF_{N2O-N} $*GWP_{n2o}$ $N_2O-N_{ST,D}$= $EF_3*TKN_D*t,_D$	TKN_D, t_D	EF_5=0.0075[17,18] $Frac_{runoff}$=0.007[18] EF_3=0.005[17,18]	$Frac_{LEACH}$=0-0.8[17] EF_1= 0.05-0.5[17]
Land Application-digestate	$EM_{LA,D}$=EM_{spread}+$EM_{N2OLA,D}$-$EM_{fert,D}$- CS_D $EM_{ST,vol,D}$= $N_{vol,ST,D}$ $*EF_4*$ $CF_{N2O-N}*GWP_{N2O}$ $N_{vol,ST,D}$= $Frac_{GASMS}$ * $TKN_D*t,_D$	TKN_D, t_D	EF_4 =0.01[17] $Frac_{GASMS}$=0.26[17,18](Table A-205)	EF_4=0.002 - 0.050[17] $Frac_{GASM}$= 0.05-0.50[17]
Digestate Land application	$EM_{LA,D}$=EM_{spread}+$EM_{N2OLA,D}$-$EM_{fert,D}$- CS_D $EM_{spread,D}$=EF_{spread} $*t_{LA,D}$	t_D	EF_{spread}=1.45[12d]	
	$EM_{spread,D}$=EF_{spread} $*t_{LA,D}$	tD	EFspread=1.45[12d]	
Land Application-direct and indirect	$EM_{LA,D}$ =$EM_{LAN2Od,D}$+$EM_{LA,vol,D}$+$EM_{LA,runoffleach,D}$			

	$EM_{N2Odirect,LA,D}= N_2O\text{-}N_{direct,LA,D} * CF_{N2O\text{-}N\text{-}N} * GWP_{N2O}$ $N_2O\text{-}N_{direct, LA,M}= EF_1 * N_{LA,D}$ $N_{LA,D}= N_D - N_2O\text{-}N_{direct,ST,D} - Nvol_{ST,D} - N_{leachrunoff,D} - (NO_{loss} - N_{2loss})_D$	t_D, TKN_D, TAN_D	$EF_1=0.0125$[17 (Table 11.1)] $NO_{loss}=0.012$ $N_{2loss}= 0.08$	
	$EM_{N2Ovol,LA,D} = N_{vol,LA,D} * EF_4 * CF_{N2O\text{-}N} * GWP_{N2O}$ $N_{vol, LA,D}= * N_{LA,D} * Frac_{GASM}$ $EM_{N2Odirect,LA,D}= N2O\text{-}N_{direct,LA,D} * CF_{N2O\text{-}N\text{-}N} * GWP_{N2O}$ $N_2O\text{-}N_{direct, LA,M}= EF_1 * N_{LA,D}$		$EF_4=0.01$ [17 (Table 11.3)] $Frac_{GASM}=0.20$[17 (Table 11.3] $EF_1=0.0125$[17 (Table 11.1)]	Same as above
	$EM_{N2Orunoffleach,D} = N_{runoffleach,D} * EF_5 * CF * GWP_{N2O}$ $N_{runoffleach,D}= Frac_{runoff,ls,MA} * N_{LA,D} N_{LA,M}= N_D - N_2O\text{-}N_{direct,ST,D} - Nvol_{ST,D} - N_{leachrunoff,D} - (NO_{loss} - N_{2loss})_D$ $N_{org,LA,M}=(TKN_D - TAN_D) * t_D - N_2O\text{-}N_{ST,M} - Nvol_{ST,D} - N_{runoof,ST,D} - NO_{loss} - N_{2loss}$	t_D, TKN_D, TAN_D,	$EF_5=0.0075$[15 (Table 11.3)] $Frac_{runoff,ls,MA}= 0.007$[16] $NO_{loss}=0.012$ $N_{2loss}= 0.08$	Same as above
Fertilizer displacement, manure land application ($EM_{fert,M}$)	$EM_{fert,D}=Neff_D * (EF_N + EF_{N2O,IF}) + Pmin_{factor} * P_D * t_D * EF_P$ $N_{effD}= N_{org,LA,D} * Min_{factor,NY} + Nmin_{LA,D} - N2O\text{-}N_{LA,M} - N_{vol,LA,M} - N_{LA,runoff,D}$	P_D, t_D	$Min_{factor,NY,M}=0.52$[14] $Pmin_{factor}=0.90$[15] $EF_N=6.8$[16] $EF_{N2O,IF}=5.4$[10] $EF_P=0.41$[16]	Same as above
Carbon Sequestration, digestate land application	$CS_{LA,D}=CS_{factor} * VS_D * t_D * CF_{C\text{-}CO2} EM_{N2Orunoffleach,D} =$	VS_D, t_D	$CS_{Dfactor}=0.10 EF_5=0.0075$[15 (Table 11.3)] $Frac_{runoff,ls,MA}= 0.007$[16]	CS_D=+/20%Same as above

Fertilizer displacement, manure land application ($EM_{fert,M}$)	$EM_{fert,D}$=$Neff_D$*(EF_N+$EF_{N2O,IF}$)+$Pmin_{factor}$*P_D*t_D* EF_P N_{effD}= $N_{org,LA,D}$*$Min_{factor,NY+}$ $Nmin_{LA,D}$–N_2O-$N_{LA,M}$-$N_{vol,LA,M}$-$N_{LA,runoff,M}$	P_D, t_D	$Min_{factor,NY,M}$=0.52[14] $Pmin_{factor}$=0.90[15] EF_N=6.8[16] $EF_{N2O,IF}$=5.4[10] EF_P=0.41[16]	

[a] Table 3-1.
[b] Table 3-4.
[c] Table 3-3
[d] Scaled based upon reported transportation distance/20km. 11/20 for manure; 19/20 for digestate

Table B- 3: Key parameters linked to land application emissions for the reference and AcoD case

	Manure $_{(i=M)}$	Influent $_{(i=IN)}$	Digestate $_{(i=D)}$	Dairy[a]
Volume (t)	88,247	120,271	115,460	26,977
Solids content prior to storage (TS_i (g/kg))	72.2	67.8	42.4	40
Volatile solids content prior to storage (VS_i (g/kg))	56.6	57.0	30.4	
Total N content prior to storage (TKN_i (mg/L))	3,540	2,827	3,097	800
Total N prior to storage N_i (kg/yr)	312,366	340,007	357,600	21,581
Ammonaical N content prior to storage (TAN_i (mg/kg))	1,623	870	1,421	11
Ammoniacal N prior to storage (kg/yr)	143,196	104,616	164,050	296
% Ammonaical N (TAN/N) prior to storage	45.8%	30.8%	45.9%	1.4%
P_i content (mg/kg) prior to storage	435	477	412	400
K_i content (mg/kg) prior to storage	3,733	1,371	1,429	35
pH_i prior to storage	6.97	6.89	7.83	4.25
Estimate Storage N loss	36%		36%	
Estimated Net Total N applied ($N_{LA,I}$) (kg/yr)	200,226	na	227,434	21,581
Estimated Net TKN(mg/kg)	2,269		1,970	
Estimated Min N applied ($MIN_{LA,I}$ (kg/yr))	125,746		134,530	
Estimated Land Application N losses (kg/yr)	35%		37%	
Estimated Effective N available (kg/yr)	81,796	na	84,608	4,012
Total N available/initial (kg/yr)	26.2%	na	23.7%	19.6%

[a] Dairy land application emissions are reported under the food disposal section of the reference case (not under the land application section). They are included here for reference and completeness.

Table B- 4 : Results of computer simulation of IPPC land application emission factor uncertainty ranges for nitrous emissions related to reference case manure land application.

Factor	Coeff. of correlation
EF_3	0.215538
$Frac_{GASMS}$	-0.00268
EF_4	0.842955
EF_1	0.277517
$Frac_{GASM}$	0.106594
$Frac_{Leach/Runoff}$	0.102767
EF_5	0.099392
$Frac_{N2/NO}$	-3.44E-15
Total Nitrous emissions (kg CO_2e/yr)[a]	
n(simulations)	2187
Min	481,178
Max	6,476,302
$Nominal_{M,NY}$	996,637

[a] Values differ slightly due to use of IPCC AR4 GWP factors. Chart not updated to AR5

Table B- 5: Lifecycle Inventory Data, emissions of greenhouse gasses per phase for the reference case and AcoD case per t influent

Contributions	fossil direct (kg CO_2t influent)	Fossil indirect (kg CO_2/t influent)	Cseq (kg C/t influent)	CH_4 (kg CH_4/t influent)	N_2O direct (kg N_2O/t influent)	N_2O indirect (Kg N_2O/t influent)
Reference Case						
Manure Storage	**-**	**-**	**-**	**1.6**	**0.0**	**0.0**
EMch4 (methane emissions)	-	-	-	1.6	-	-
EM n2od (direct N_2O emissions)	-	-	-	-	0.0	-
EMn20o vol (indirect N_2O volatilization)	-	-	-	-	-	0.0
EMn2o (indirect N_2O runnoff)	-	-	-	-	-	0.0
Net Manure Land Application	0.0	(0.0)	(0.0)	-	0.0	0.0
Manure Land Application	0.6	-	-	-	0.0	0.0
EMLA,spreader	0.6	-	-	-	-	-
EMn2odirect	-	-	-	-	0.0	-
EMLA, indrect n2o vol	-	-	-	-	-	0.0
EMLA,indirect n2o leach	-	-	-	-	-	0.0
EMLA, fertilizer displacement	-	(7.9)	-	-	-	-
EMLA, CS	-	-	(5.4)	-	-	-
FW Disposal Emissions	**0.0**	**(0.0)**	**(0.0)**	**0.0**	**0.0**	**-**
EMWWT/LF	1.8	(2.3)	(0.1)	0.6	-	-
EMwwlf, DAF	-	-	-	-	-	-
Emxport, DAF	0.1	-	-	-	-	-
EMCH4LF, DAF	-	-	-	0.1	-	-
EMOP,DAF	0.9	-	-	-	-	-
Emgrid, DAF	-	(0.4)	-	-	-	-
EMCS, DAF	-	-	(0.0)	-	-	-
EMwwlf, GTW	-	-	-	-	-	-
Emxport, GTW	0.1	-	-	-	-	-
$EMCH_4LF$, GTW	-	-	-	0.5	-	-
EMOP,GTW	0.7	-	-	-	-	-
Emgrid, GTW	-	(1.9)	-	-	-	-
EMCS, GTW						

	-	-	(0.1)	-	-	-
EMWWT	1.1	-	-	-	-	-
EFMLA	2.7	(0.4)	(0.9)	0.0	0.0	-
EMLA,xport	2.4	-	-	-	-	-
EMLA, spreader	0.3	-	-	-	-	-
EMLA, direct n2o	-	-	-	0.0	-	-
EMLA, indirect n2o vol	-	-	-	-	0.0	-
EMLA, indirect n2o leach	-	-	-	-	0.0	-
EMLA, fertilizer displacement	-	(0.4)	-	-	-	-
EMLA, carbon sequestration	-	-	(0.9)	-	-	-
EMLF, FPW	0.1	(0.0)	(0.0)	0.0	-	-
ACD case						
FW Hauling	**1.1**	**-**	**-**	**-**	**-**	**-**
Digester	**-**	**(35.3)**	**-**	**0.7**	**0.0**	**-**
EMLeak	-	-	-	0.4	-	-
EMIC	-	-	-	0.3	0.0	-
EMFGRID	-	(35.3)	-	-	-	-
Storage	**-**	**-**	**-**	**1.1**	**0.0**	**0.0**
EMch4	-	-	-	1.1	-	-
EMn2odirect	-	-	-	-	0.0	-
EMn2ovol	-	-	-	-	-	0.0
Land Application	**1.4**	**(8.2)**	**(3.5)**	**-**	**0.0**	**0.0**
EMxport and spread	1.4	-	-	-	-	-
EMLA,n2odirect	-	-	-	-	0.0	-
EMLA, inderect n2o vol	-	-	-	-	-	0.0
EMLA, indirect n2o leach	-	-	-	-	-	0.0
EMLA fertilizer displacement	-	(8.2)	-	-	-	-
CSLA,D Carbon Sequestration	-	-	(3.5)	-	-	-

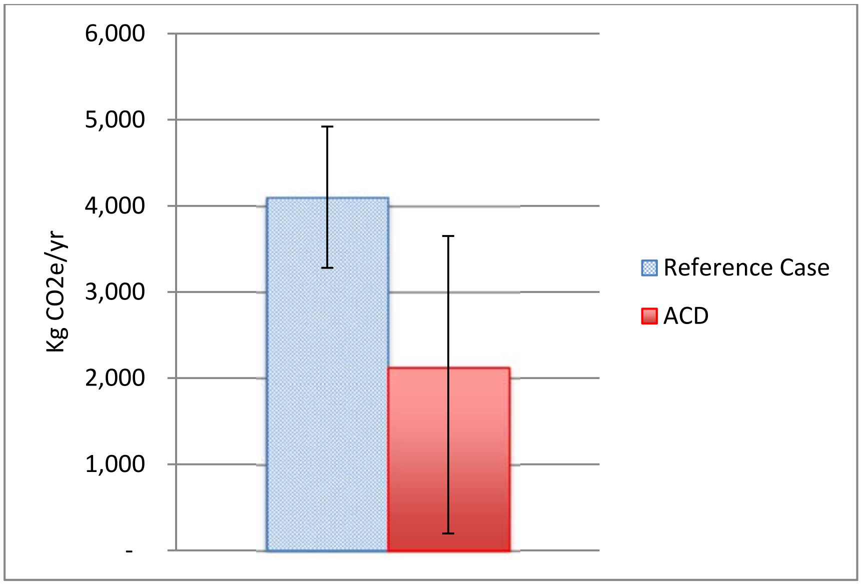


Figure B-1: Uncertainty analysis of storage phase CH_4 GHG impacts (kg CO_2e/yr) for reference and AcoD case. Reference case uncertainty is based upon IPCC uncertainty range of +-20% for a country specific MCF. AcoD uncertainty is the range of empirical data reported (0.004-0.074) in Table B-2.

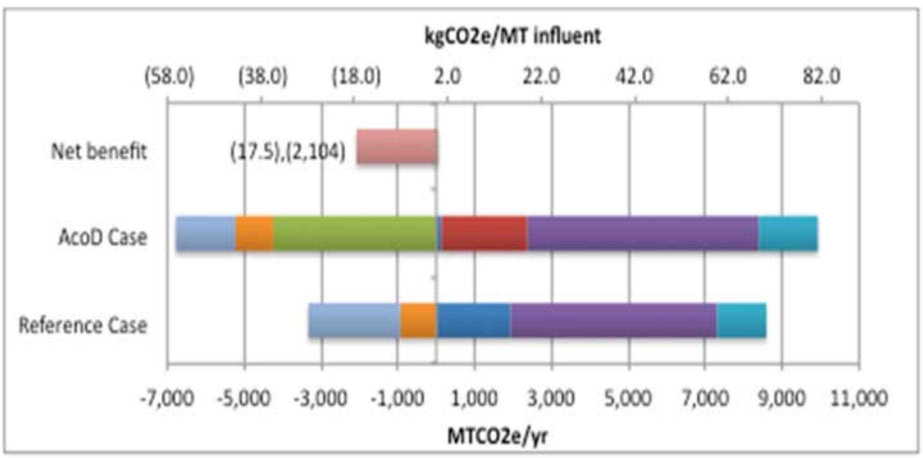


Figure B-2: Scenario analysis of storage phase CH uncertainty. Reference case $MCF_{ls,ny}$ =0.192 (EPA regional factor +20% uncertainty per IPCC protocol)and Digester storage emissions = 0.074m3CH_4/kgVS based upon high values of reported empirical range. (MT=metric tons)

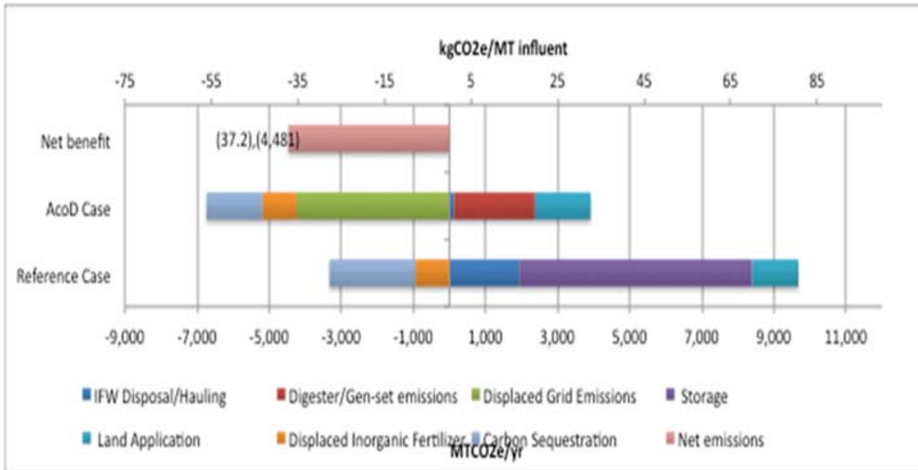


Figure B-3: Sensitivity analysis to fertilizer displacement; assumes inorganic fertilizer application is unchanged between reference case and AcoD case. (MT= metric tons)

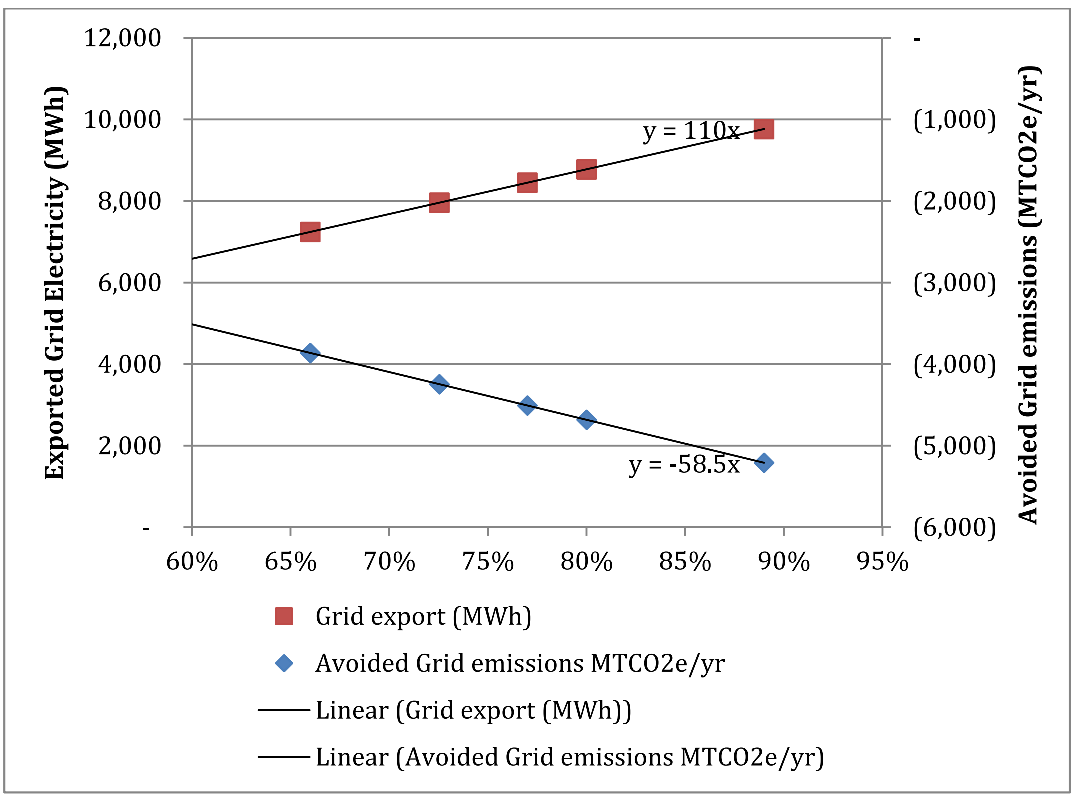


Figure B-4: Sensitivity analysis of Capacity factor to electricity exported and avoided grid emissions based upon the NPCC regional grid mix.

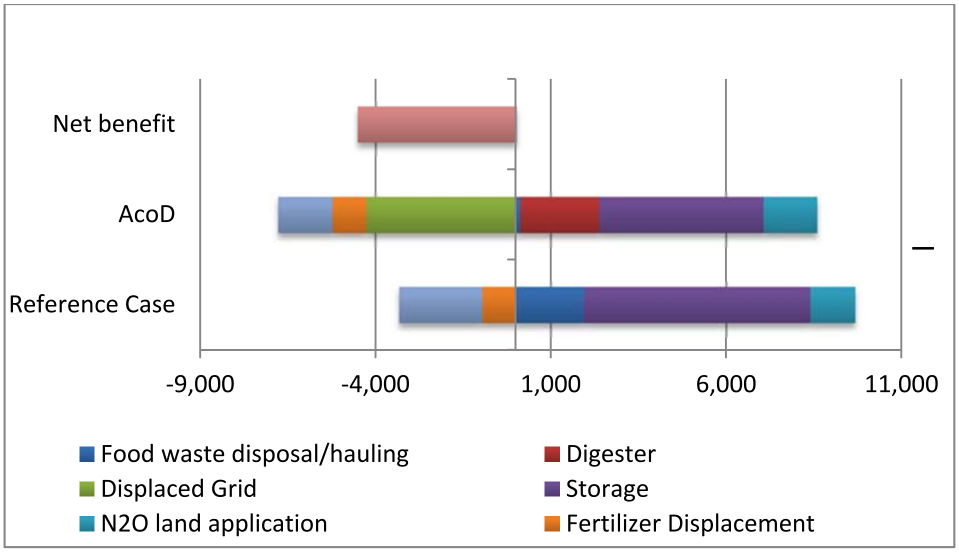


Figure B-5: Sensitivity analysis of grid emission factor, applying national grid mix emission factor for LFG recovery and AD grid displacement. Net benefit 4,512 tCO_2e/t (37.5g CO_2e/t influent)

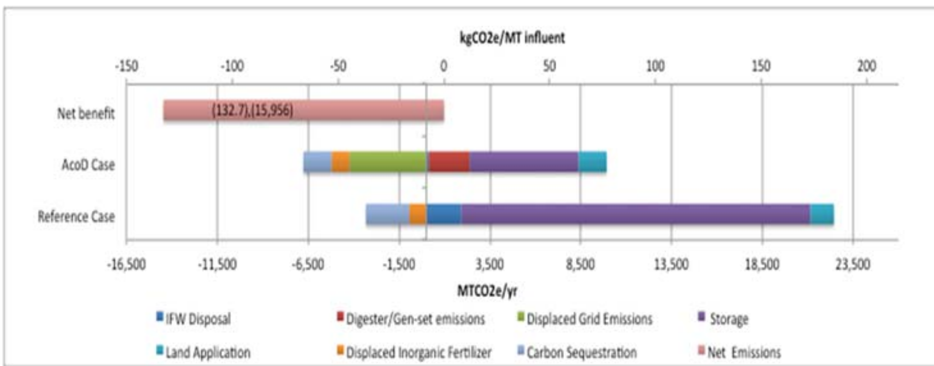


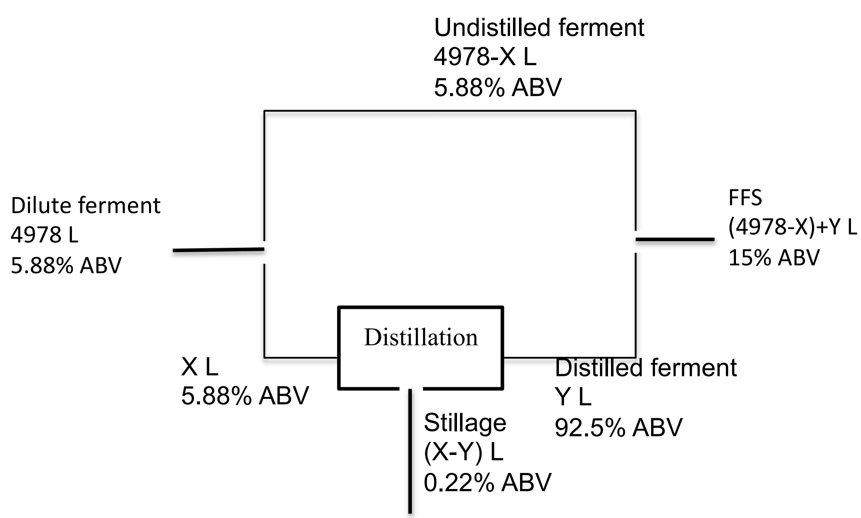
Figure B-6: Sensitivity analysis to climate and storage technique; MCF_M=0.8 (Anaerobic lagoon in Florida); $EMCH_{4,D}$=0.074m3CH_4/gVS (upper reported value)

Equation B-1: Mass balance to Phase 1.5 Concentration and volumes

4978 L enters the process at 5.88% ABV. Some portion (X) is distilled resulting in an amount (Y) at 92.5% and sillage (X-Y) at 0.22%ABV.

The distilled ferment (YL at 92.5%ABV) is blended with undistilled ferment ((4978-X)L at 5.88%ABV) to achieve a fuel feed slurry (FFS) at 15% AVB.

Both ethanol and ferment are mass balanced assuming constant densities.



Ethanol Balance:

Undistilled fermented ethanol content plus distilled ethanol content equals FFS ethanol content

$$(4978 - x)(0.0588) + y((.925) = (4978 - x + y)(.15)$$

$$454 = .775y + 0.912x \qquad (1)$$

Ethanol content into distillation equals ethanol content in distilled ferment plus ethanol loss in stillage

$$0.0588x = .925y + (x - y)(0.0022)$$

$$x = 16.3y \qquad (2)$$

substituting (2) into (1)

y = 201 L distilled to 92.5% ABV (186 L anhydrous ethanol)

x = 3274 L

4978 – x = 1704 L undistilled ferment at 5.88% ABV

4978 – x + y = 1904 L FFS

x – y = 3074 L stillage at 0.22% ABV

Equation B-2: Mass balance to calculate animal feed co-product

Calculation of Animal Feed Co-product

$$Dry\ solids = ethanol + CO2 + dry\ mass\ of\ compost + dry\ mass\ of\ DDGS$$

Equation B-3: Calculation of theoretical yield

Calculation of theoretical yield

Co-fermentation feedstock consists of 4720 kg total mass input: 2310 kg diluent and 2410 kg food scraps.

The solids content of the diluent (2410 kg wet mass @ 10% solids = 241 kg DM) was assumed to consist of 50% fructose and 50% glucose.

Food scraps (2310 kg wet mass @ 30% solids = 693 kg DM) were assumed to have be similar in composition to the synthetic food scraps analyzed by Schmitt et al. (2012): arabinose, 0.9%; galactose, 0.3%; glucose, 65.1%; xylose, 7.9%; mannose, 4.5%.

*Specific gravity at 20°C

Appendix C

Table C 1: Summary table containing bio-methane yield (B_o) for the substrates tested (mL CH_4/g VS) shown in red with axis below graph. Methane yield per unit mass (Lo) (m^3 CH_4/t_{FW}) shown in blue with axis above graph. Substrates were tested in triplicate (n=3) unless otherwise noted. Error bars represent one standard deviation (σ).

Substrate	Bio-methane potential Bo	Extent of bio-degradation	Apparent hydrolysis rate coefficient
	($mLCH_4/gVS$)a	f_d ($mLCH_4/mLCH_4$)[a]	k_h (d^{-1})[b]
Cellulose (C) (n=15)	353(44)	0.85 (0.11)	0.32(0.032)
Manure (M) (n=12)	**238 (19)**	**0.54 (0.04)**	**0.19 (0.111)**
Baked goods (BG)	465(26)	0.94 (0.05)	0.26(0.007)
Canned goods (CG)[c]	436 (10)	0.98 (0.02)	0.32 (NA)[h]
Coffee/filter paper (COF) (n=6)	365(57)[d]	0.80 (0.13)	0.14 (0.009)
Fruit and Veg Waste (FVW)[e]	418 (58)	0.98 (0.14)	0.34 (0.010)
Soiled napkins (N) (n=2)	382(59)	0.91 (0.14)	NA[h]
Post-consumer (POST) (n=6)	483(86)[f]	0.88 (0.16)	0.27(0.016)
Kitchen prep waste (PREP) (n=9)	252(40)	0.56 (0.09)	0.48 (0.027)
Sweet dry goods (SDG)	362(36)	0.84 (.08)	0.20 (0.003)
Salad mix (SM)	375 (21)	0.90 (0.05)	0.64 (0.049)
Unsweetened dry goods (UDG) (n=6)	318(86)	0.74 (0.20)	0.47 (0.033)
Yogurt/Frozen desserts (YFD)[g]	454 (6)	0.99 (0.01)	0.45 (0.059)
Cafe blend (CAFE) (n=6)	475(32)	0.98 (0.07)	0.38 (0.011)
Food service blend(SERVICE)	496(12)	0.91 (0.02)	0.28 (0.015)
Retail blend (RETAIL((n=9)	462(37)	0.99 (0.08)	0.42 (NA)[h]
Baked goods:manure (BG:M)	437(12)	0.90 (0.02)	0.27 (NA)[h]
Canned goods:manure (CG:M) (n=6)	362(53)	0.82 (0.12)	0.27(0.007)
Fruit/Veg waste:manure (FVW:M)	308 (91)	0.71 (0.21)	0.19 (0.005)
Kitchen Prep:manure (PREP:M)	165(23)	0.37 (0.05)	0.35(0.014)
Post-consumer:manaure (POST:M)	344(33)	0.67 (0.06)	NA[h]
Retail blend:manure (RETAIL:M) (n=6)	374(62)	0.82 (014)	0.44 (0.019)
Sweet dry goods:manure (SDG:M)	325(26)	0.74 (0.06)	0.25 (0.011)
Food service blend:manure (SERVICE:M)	466(47)	0.92 (0.09)	0.30 (0.011)
Unsweetened dry goods:manure (UDG:M) (n=8)	372(42)	0.86 (0.10)	0.41 (0.023)

Appendix D

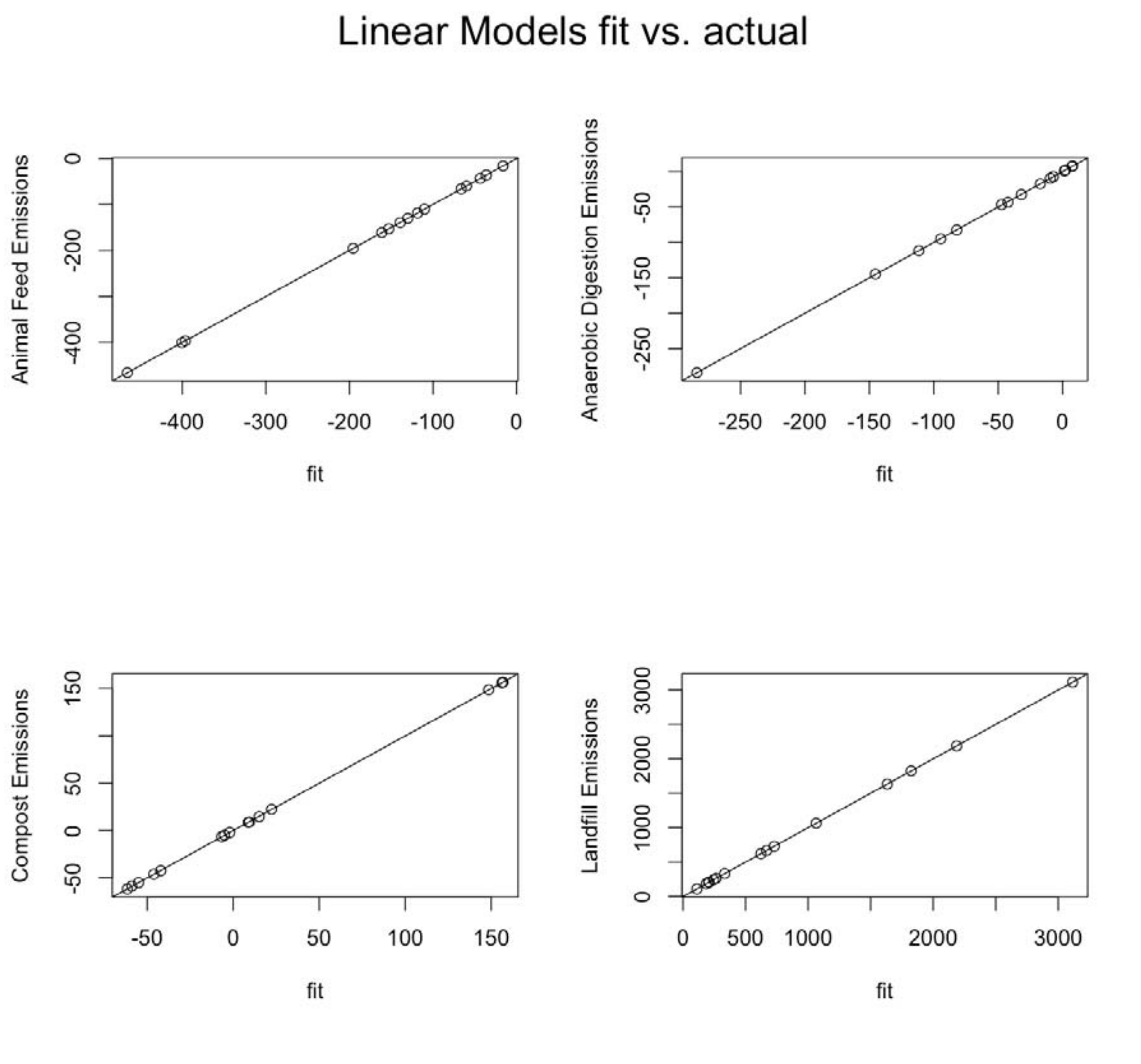


Figure D-1: Linear fit regressions

Regression statistics:

Animal Feed

Residuals:

Min	1Q	Median	3Q	Max
-7.670e-14	-2.049e-14	-2.301e-15	9.106e-15	1.039e-13

Coefficients:	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	5.871e-14	1.675e-14	3.504e+00	0.00388 **
TS:TDN	-6.880e+02	5.351e-14	-1.286e+16	< 2e-16 ***

Residual standard error: 4.128e-14 on 13 degrees of freedom

Multiple R-squared: 1, Adjusted R-squared: 1

F-statistic: 1.653e+32 on 1 and 13 DF, p-value: < 2.2e-16

Scaled Animal feed

(Intercept)	scaledTS_TDN
-16.24889	-449.48601

Anaerobic digestion

Residuals:

Min	1Q	Median	3Q	Max
-0.89204	-0.21772	-0.00017	0.19642	0.73827

Coefficients:	Estimate	Std. Error	t value Pr(>|t|)
(Intercept)	8.110e+00	2.002e-01	40.51 2.01e-12 ***
Lo	-1.349e+00	3.909e-03 -	345.15 < 2e-16 ***
TVS	4.619e+02	3.778e+00	122.25 < 2e-16 ***
TKN	1.179e-03	5.276e-05 22.35 7.22e-10 ***	
InitialC -	3.683e-01	6.257e-03 -58.87 4.86e-14 ***	

Residual standard error: 0.479 on 10 degrees of freedom

Multiple R-squared: 1, Adjusted R-squared: 1

F-statistic: 9.454e+04 on 4 and 10 DF, p-value: < 2.2e-16

Scaled AD

(Intercept)	scaledLo	scaledTVS	scaledTKN	scaledInitialC
1.149478	-545.569077	401.478419	20.382547	-154.418641

Compost

Residuals:

Min	1Q	Median	3Q	Max
-0.44230	-0.07717	-0.01967	0.14404	0.34734

Coefficients:	Estimate Std.	Error t	value	Pr(>|t|)
(ntercept)	-7.236e+01	1.065e-01	-679.7	<2e-16 ***
npert	2.714e+00	2.303e-02	117.8	<2e-16 ***
InitialC	4.351e-01	9.744e-04	446.6	<2e-16 ***

Residual standard error: 0.2583 on 12 degrees of freedom

Multiple R-squared: 1, Adjusted R-squared: 1

F-statistic: 6.178e+05 on 2 and 12 DF, p-value: < 2.2e-16

Scaled Compost:

(Intercept)	scalednpert	scaledInitialC
-64.07409	46.90354	182.42441

Landfill:

Residuals:

Min	1Q	Median	3Q	Max
-2.509e-13	-1.114e	-13 -1.873e	-14 3.391e	-14 4.981e-13

Coefficients:	Estimate Std.	Error	t value	Pr(>|t|)
(Intercept)	1.848e+01	8.118e	-14 2.277e+14	<2e-16 ***
Lo	7.650e+00	4.880e-16	1.568e+16	<2e-16 ***
InitialC:rdeg	-3.667e+00	1.808e-15	-2.028e+15	<2e-16 ***

Residual standard error: 2.028e-13 on 12 degrees of freedom

Multiple R-squared: 1, Adjusted R-squared: 1

F-statistic: 1.384e+32 on 2 and 12 DF, p-value: < 2.2e-16

Scaled Landfill:

(Intercept)	scaledLo	scaled(InitialC *rdeg)
90.15754	3737.96103	-790.45787

Linear models simply based upon TS.

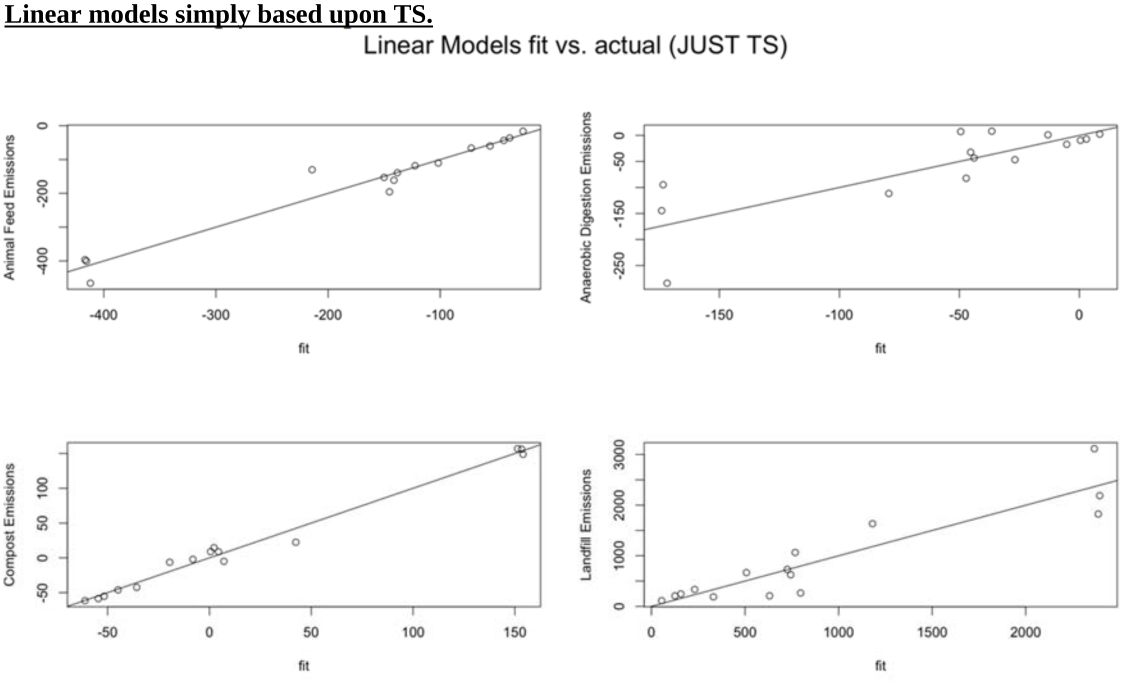


Figure D-2: Linear regression base only on TS fit

Animal feed based only on TS

Coefficients:	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-9.478	12.952	-0.732	0.477
TS	-439.402	27.630	-15.903	6.7e-10 ***

Residual standard error: 32.55 on 13 degrees of freedom

Multiple R-squared: 0.9511, Adjusted R-squared: 0.9474

F-statistic: 252.9 on 1 and 13 DF, p-value: 6.698e-10

AD based on TS only

Coefficients: Estimate Std. Error t value Pr(>|t|)

(Intercept)	16.27	18.55	0.877	0.396300
TS	-205.32	39.57	-5.189	0.000174 ***

Residual standard error: 46.62 on 13 degrees of freedom

Multiple R-squared: 0.6744, Adjusted R-squared: 0.6494

F-statistic: 26.93 on 1 and 13 DF, p-value: 0.0001743

Compost based just on TS:

Coefficients:	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-70.316	3.719	-18.91	7.70e-11 ***
TS	241.975	7.933	30.50	1.76e-13 ***

Residual standard error: 9.346 on 13 degrees of freedom

Multiple R-squared: 0.9862, Adjusted R-squared: 0.9852

F-statistic: 930.4 on 1 and 13 DF, p-value: 1.755e-13

Landfill based just on TS:

Coefficients:	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	-45.64	146.61	-0.311	0.761
TS	2632.91	312.76	8.418 1	.27e-06 ***

Residual standard error: 368.5 on 13 degrees of freedom

Multiple R-squared: 0.845, Adjusted R-squared: 0.8331

F-statistic: 70.87 on 1 and 13 DF, p-value: 1.274e-

Table D-1: One dimensional sensitivity analysis summary

Treatment pathway parameter	Apple Pomace (AP)	Baked goods (BG)	Brewers spent grains (BS)	Canned goods (CG)	Coffee grounds and paper (COF)	Dry goods (DG)	Fresh produc e (FP)	MSWFW	Post consumer (POST)	Prep waste (PRE P)	Refrig.d and frozen goods (RFG)	Salad (S)	Sweet Cereals (SC)	Tomat o Pomac e (TP)	Whey
Landfill															
EFGrid	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%	14%
LFDieseluseLpert	12%	2%	7%	15%	6%	2%	21%	7%	3%	18%	5%	49%	2%	9%	26%
Landfill_OX_Max	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%	-21%
Landfill_CF	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%	3%
LCEMax	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%
BMP_Correctionfactor	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%	92%
k	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Max uncertainty	131	1038	221	103	265	716	75	225	544	90	342	32	794	170	62
Nominal	208	3115	668	333	730	1824	246	623	187	1633	1064	111	2188	264	207
Animal Feed															
loss	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	
Nominal	-118	-465	-110	-59	-139	-400	-44	-161	-66	-130	-195	-16	-396	-153	-22%
Max variation	59	233	55	30	70	200	22	81	65	33	98	8	198	76	-5%
Compost															-12%
Compost_degraded C_CH4	-19%	-68%	-16%	-7%	-22%	-65%	-5%	-23%	-21%	-10%	-22%	-3%	-63%	-22%	-1%
Compost_dieseLlpert	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-5%	-14%
Compost_electpert	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-12%	-3%
Compost_N_loss	-1%	-3%	-1%	0%	-1%	-3%	0%	-2%	-2%	0%	-1%	0%	-3%	-1%	100%
Compost_N2OperN	-12%	-27%	-15%	-5%	-14%	-32%	-2%	-16%	-24%	-1%	-13%	-3%	-30%	-14%	-11%
Compost_NH3ofloss	-3%	-6%	-3%	-1%	-3%	-7%	-1%	-4%	-6%	0%	-3%	-1%	-7%	-3%	18%
Compost_Peat_Displa cement	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	0%
CompostPercentCdegr aded	-10%	-36%	-8%	-4%	-11%	-34%	-3%	-12%	-11%	-6%	-12%	-1%	-34%	-11%	17%
EF_Peat_kgCO2eperto n	18%	18%	18%	18%	18%	18%	18%	18%	18%	18%	18%	18%	18%	18%	-388

EFGrid	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	
Peat_substitution	17%	17%	17%	17%	17%	17%	17%	17%	17%	17%	17%	17%	17%	17%	-24%
Max impact	-388	-388	-388	-388	-388	-388	-388	-388	-388	-388	-388	-388	-388	-388	-8%
Anaerobic Digestion															-32%
AD_Cf	-17%	-24%	-24%	-24%	-24%	-24%	-24%	-24%	-24%	-19%	-24%	-24%	-24%	-16%	100%
AD_CSfactor	-29%	-7%	-10%	-8%	-14%	-18%	-8%	-22%	4%	-23%	-7%	-13%	-13%	-26%	13%
AD_Digester_CE	-23%	-32%	-32%	-32%	-32%	-32%	-32%	-32%	-32%	-26%	-32%	-32%	-32%	-22%	15%
AD_Digester_CH_4Leaks	73%	100%	100%	100%	100%	100%	100%	100%	100%	81%	100%	100%	100%	70%	0%
AD_Digester_parasiticLoad	9%	13%	13%	13%	13%	13%	13%	13%	13%	10%	13%	13%	13%	9%	4%
AD_flared	11%	15%	15%	15%	15%	15%	15%	15%	15%	12%	15%	15%	15%	11%	-2%
AD_LA_FracGasD	1%	0%	1%	1%	1%	1%	1%	1%	1%	0%	1%	1%	1%	0%	-38%
AD_LA_FracLeachD	14%	5%	13%	9%	11%	9%	6%	15%	9%	2%	7%	16%	7%	2%	4%
AD_N_Availability	-9%	-3%	-8%	-6%	-7%	-5%	-4%	-9%	-6%	-1%	-5%	-10%	-4%	-1%	3%
AD_reductionInVS	-51%	-35%	-36%	-37%	-44%	-50%	-38%	-48%	-33%	-51%	-35%	-43%	-44%	-51%	74%
AD_Storage_EF3	15%	6%	15%	10%	12%	10%	7%	16%	10%	2%	8%	18%	8%	2%	38%
AD_Storage_FracGasMS	10%	4%	9%	7%	8%	7%	4%	11%	7%	1%	5%	12%	5%	1%	0%
AD_Storage_residual CH_4	100%	67%	70%	72%	86%	98%	75%	94%	65%	100%	69%	83%	86%	100%	-2%
AF_loss	43%	29%	33%	38%	34%	36%	38%	47%	16%	39%	37%	33%	33%	41%	0%
Displaced_K_Production_Factor	-1%	0%	0%	0%	-1%	0%	0%	-1%	0%	0%	0%	-1%	0%	0%	-61%
Displaced_N_Production_Factor	-8%	-3%	-8%	-5%	-7%	-5%	-4%	-9%	-5%	-1%	-4%	-10%	-5%	-1%	7%
Displaced_P_Production_Factor	0%	0%	-1%	0%	-1%	0%	0%	-1%	0%	-1%	0%	-1%	0%	0%	0%
EFGrid	0%	-61%	-61%	-61%	-61%	-61%	-61%	-61%	-61%	-50%	-61%	-61%	-61%	-43%	0%
IPCC_EF4	26%	10%	25%	17%	20%	17%	11%	27%	17%	3%	14%	29%	14%	3%	0%
LandApplication_EF1	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	-2%
MF_N2O	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	47
MF_ROL	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	
N_displacement	-9%	-3%	-8%	-6%	-7%	-6%	-4%	-9%	-6%	-1%	-5%	-10%	-5%	-1%	
Max impact	138	797	170	79	203	549	57	172	418	85	262	25	608	187	14%

References

Abdollahzadeh, F., Pirmohammadi, R., Fatehi, F., & Bernousi, I. 2010. Effect of feeding ensiled mixed tomato and apple pomace on performance of Holstein dairy cows. Slovak J Anim Sci, 43, 31-35.

Aden, A., Ruth, M., Ibsen, K., Jechura, J., Neeves, K., Sheehan, J., and Wallace, R. Lignocellulosic biomass to ethanol process design and economics utilizing co-current dilute acid prehydrolysis and enzymatic hydrolysis for corn stover. NREL/TP-510-32438. National Renewable Energy Laboratory, Golden, CO; 2002.

Amon, A.; Kryvoruchko, V.; Amon, T.; Zechmeister-Boltenstern, S. 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence or slurry treatment. Bioresource Technology. 112, 153-162.

Amon, R., Jenner, M., El-Mashad, H., Williams, R., Kafka, S., Public Interest Energy Research (PIER) Program FINAL PROJECT REPORT□□California Food Processing Industry Organic Residue Assessment, California biomass collaborative,
APRIL 2012, http://www.energy.ca.gov/2013publications/CEC-500-2013-100/CEC-500-2013-100.pdf

American Recycler News Inc (ANRI), “More states ban organic waste in landfills”, January 2014, http://www.americanrecycler.com/0114/2428more.shtml

Argonne National Lab (ANL), The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model, GREET 2015 release, https://greet.es.anl.gov/

Andersen, J. K., 2010. Composting of organic waste: quantification and assessment of greenhouse gas emissions (Doctoral dissertation, PhD Dissertation Technical University of Denmark, Lyngby, Denmark)
Angelidaki, I., & Sanders, W. 2004. Assessment of the anaerobic biodegradability of macropollutants. Reviews in Environmental Science & Biotechnology, 3(2), 117-129.

Angelidaki, I, Alves, M., Bolzonella, D., Borzacconi, L., Campos, J.L., Guwy, A.J., Kalyuzhnyi, S., Jenicek, P., van Lier, J.B., 2009. Defining the biomethane potential (BMP) of solid organic wastes and energy crops: a proposed protocol for batch assays. Water Sci. Technol., 59(5), 927–934.

Astals, S., Batstone, D. J., Mata-Alvarez, J., Jensen, P. D. 2014. Identification of synergistic impacts during anaerobic co-digestion of organic wastes. Bioresource Technology, 169, 421-427

ASTM Standard E2170 – 01, 2008. Standard test method for determining anaerobic biodegradation potential of organic chemicals under methanogenic conditions. ASTM International, West Conshohocken, PA.

(AOAC) Official Methods of Analysis of AOAC INTERNATIONAL (2012) 19th Ed., AOAC INTERNATIONAL, Gaithersburg, MD, USA, 984.13.

(AOAC) Official Methods of Analysis of AOAC INTERNATIONAL (2012) 19th Ed., AOAC INTERNATIONAL, Gaithersburg, MD, USA, 991.36.

(AOAC) Official Methods of Analysis of AOAC INTERNATIONAL (2012) 19th Ed., AOAC INTERNATIONAL, Gaithersburg, MD, USA, 986.25.

APHA,AWWA,WPCF, (1998) Standard Methods for the Examination of Water,Wastewater and Sludge. Washington, D.C.

Artrip, K.; Shrestha, D.; Coats, E.; Keiser, D. 2013. GHG emissions reduction from an anaerobic digester in a eairy farm: Theory and practice. Appl. Eng. Agric. 29(5), 729-737.

Baily, R. Anaerobic digestion of restaurant grease wastewater to improve methane gas production and electrical power generation potential. Proceedings of the Water Environment Federation, 2009, WEFTEC 2007: Session 81 through Session 90, 6793-6805.

Baky, A. & Eriksson, O., 2003. Systems analysis of organic waste management in Denmark. Environmental Project No. 822. Danish Environmental Protection Agency, Copenhagen, Denmark.

Bartram, D. and Barbour, W. Estimating greenhouse gas reductions for a regional digester treating dairy manure. Paper presented at the 13th International Emission Inventory Conference, Clearwater, FL., June 9, 2004.

Banks, C. J., Chesshire, M., Heaven, S., & Arnold, R. 2011. Anaerobic digestion of source-segregated domestic food waste: performance assessment by mass and energy balance. Bioresource technology, 102(2), 612-620.

Barlaz, M.A. 1998. Carbon Storage during Biodegradation of Municipal Solid Waste Components in Laboratory-Scale Landfills; Global Biogeochem. Cycles, 12, 373-380.

Batstone, D. J., Keller, J., Angelidaki, I., Kalyuzhnyi, S. V., Pavlostathis, S. G., Rozzi, A. and Vavilin, V. A. 2002. The IWA Anaerobic Digestion Model No 1(ADM 1). Water Science & Technology, 45(10), 65-73.

BDI, Maier,Thomas, Digestion of Brewery Residues –Potential and Challenges, Final Conference FABbiogas -2 September 2015, Brussels, http://european-biogas.eu/fabbiogas-final-conference/

Beck-Friis, B., Pell, M., Sonesson, U., Jönsson, H., & Kirchmann, H. 2000. Formation and emission of N_2O and CH_4 from compost heaps of organic household waster. Environmental Monitoring and assessment, 62(3), 317-331.

BEEF magazine, "2015 Feed Composition Table", http://beefmagazine.com/site-files/beefmagazine.com/files/uploads/2015/02/2015-BEEF-Magazine-Feed-Comp-Tables.pdf

Bentley, C.; Gooch, C. A.; Scott, N. R. Greenhouse gas emissions from a community anaerobic digester with mixed organic wastes. 2010, Paper presented at the 2010 ASABE Annual International Meeting. Pittsburgh, PA. June 20 – June 23, 2010.

Bernstad,A.,. Jansen, J., 2012. Review of comparative LCAs of food waste management systems – current status and potential improvements", Waste Manage., 32, pp. 2439–2455

Bellmer, D. and Atieh, H. 2012. Soft drink waste for ethanol production resource. ProQuest Research Library. 19(2); 10–11.

Block, Dave, Biocycle, School district supplies organics to commercial composter,BioCycle; Aug 2000; 41, 8; ProQuestpg. 57

Boldrin, A., Andersen, J. K., Møller, J., Favoino, E., & Christensen, T. H. 2009. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Management & Research.

Boldrin, A. et al. 2011. Modelling of environmental impacts from biological treatment of organic municipal waste in EASEWASTE. Waste management31.4: 619-630.

BSR, Food Waste: Tier 1 Assessment, March 2012, http://www.foodwastealliance.org/wp-content/uploads/2013/06/FWRA_BSR_Tier1_FINAL.pdf

Börjesson, P. and Berglund, M. 2006. Environmental systems analysis of biogas systems—Part I: Fuel-cycle emissions. Biomass and Bioenergy. 30(5), 469-485.

Börjesson, P. and Berglund, M. Environmental systems analysis of biogas systems—Part II: The environmental impact of replacing various reference systems. Biomass and Bioenergy. 2007, 31(5), 326-344.

Bruun, S.; Hansen, T.L.; Christensen, T.H.; Magid, J.; Jensen, L.S. 2006. Application of processed organic municipal solid waste on agricultural land - a scenario analysis. Environmental Modeling and Assessment. 11, 251-265.

Buswell EG & Neave SL, 1930. Laboratory studies of sludge digestion. Illinois Division of State Water Survey, Bulletin No. 30. 1930, Urbana, Illinois.

Buzby, J.C., Farah-Wells, H. and Hyman, J., 2014. The estimated amount, value, and calories of postharvest food losses at the retail and consumer levels in the United States. USDA-ERS Economic Information Bulletin, (121).

Cabbai, V., Ballico, M., Aneggi, E., & Goi, D., 2013. BMP tests of source selected OFMSW to evaluate anaerobic codigestion with sewage sludge. Waste Management, 33(7), 1626-1632.

Calrecycles, Statewide Waste Characterization Study: Results and Final Report, Prepared by Cascadia, December 1, 1999, http://www.calrecycle.ca.gov/publications/Documents/LocalAsst/34000009.pdf

Calrecycles, Targeted Statewide Waste Characterization Study: Waste Disposal and Diversion Findings for Selected Industry Groups, Prepared by Cascadia, June 2006, http://www.calrecycle.ca.gov/Publications/Documents/Disposal/34106006.pdf

Calrecycles, California 2008 Statewide Waste Characterization Study, Prepared by Cascadia, August 2009, http://www.calrecycle.ca.gov/Publications/Documents/General/2009023.pdf

Calrecyles, 2014 Generator-Based Characterization of Commercial Sector Disposal and Diversion in California, Prepared by Cascadia, September 10, 2015, http://www.calrecycle.ca.gov/Publications/Documents/1543/20151543.pdf

Chan, W.H., Ebner, J., Ramchandra, R. and Trabold, T., 2013, July. Analysis of food waste resources available for sustainable energy production in the Finger lakes region of New York State. In ASME 2013 7th International Conference on Energy Sustainability collocated with the ASME 2013 Heat Transfer Summer Conference and the ASME 2013 11th International Conference on Fuel Cell Science, Engineering and Technology (pp. V001T13A006-V001T13A006). American Society of Mechanical Engineers.

Chanton, J.R.; Powelson, D.K.; Green, R.B. 2009. Methane Oxidation in Landfill Cover Soils, is a 10% Default Value Reasonable? J. Environ. Qual. 38, 654-663.

Chase, L.E.; Overton, T. R.; Balbian, D.R. 2013. Using liquid whey in dairy rations. Dairy nutrition fact sheet September 2013 (Revised). 2013, Cornell Cooperative Extension.

Chester, M. and Martin, E. Cellulosic ethanol from municipal solid waste: a case study of the economic, energy, and greenhouse gas impacts in California. 2009. Environ. Sci. Technol. 43(14); 5183–5189.

Clemens, J.; Trimborn, M.; Weiland, P.; Amon, B. 2006. Mitigation of greenhouse gas emissions by anaerobic digestion of cattle slurry. Agriculture, Ecosystems & Environment. 112 (2), 171-177.

Christensen, T. H., Simion, F., Tonini, D., & Møller, J. 2009. Global warming factors modelled for 40 generic municipal waste management scenarios. Waste Management & Research.

Cho, Han Sang, Hee Sun Moon, and Jae Young Kim. 2012. Effect of quantity and composition of waste on the prediction of annual methane potential from landfills. Bioresource technology 109: 86-92.

Climate Action Reserve (CAR). Organic Waste Digestion, Project Protocol, v.2.1, 2011, Climate Action Reserve, http://www.climateactionreserve.org/how/protocols/organic-waste-digestion/

Connecticut Department of Environmental Protection (CDEP), 2001. Identifying, Quantifying, and Mapping Food Residuals from Connecticut Businesses and Institutions: An Organics Recycling Planning Tool Using GIS, prepared by Draper/Lennon

Cornell University, College of Agriculture and Life Sciences, Fertilizers for corn, web 2015, http://fieldcrops.cals.cornell.edu/corn/fertilizers-corn

Davidsson, A.; Lovstedt, C.; la Cour Jansen, J.; Gruvberger, C.; Aspegren, H. 2008. Co-digestion of grease trap sludge and sewage sludge. Waste Management. 28, 986–992.

Davis, R. Parameter estimation for simultaneous saccharification and fermentation of food waste into ethanol using Matlab Simulink. 2008. Appl. Biochem. Biotechnol. 147(1–3); 11–21.

De Guardia, A., Mallard, P., Teglia, C., Marin, A., Le Pape, C., Launay, M., ... & Petiot, C. (2010). Comparison of five organic wastes regarding their behaviour during composting: Part 1, biodegradability, stabilization kinetics and temperature rise. Waste Management, 30(3), 402-414.

De la Cruz, F. B. and Barlaz, M.A. 2010. Estimation of waste component specific landfill decay rates using laboratory-scale decomposition data. Env. Sci. Technol. 44, 4722 – 28.

De Wit, J. N. 2001. Lecturer's handbook on whey and whey products.European Whey Products Association. Bruksela, Belgia.

Dhillon, G. S., Kaur, S., Brar, S. K., & Verma, M. 2012. Potential of apple pomace as a solid substrate for fungal cellulase and hemicellulase bioproduction through solid-state fermentation. Industrial Crops and Products,38, 6-13.

Dinuccio, E., Balsari, P., Gioelli, F., & Menardo, S. 2010. Evaluation of the biogas productivity potential of some Italian agro-industrial biomasses.Bioresource Technology, 101(10), 3780-3783.

Dinsdale, R. M., Hawkes, F. R., & Hawkes, D. L. 1996. The mesophilic and thermophilic anaerobic digestion of coffee waste containing coffee grounds. Water Research, 30(2), 371-377.

Dinuccio E; Balsari, P.; Gioelli, F.; Menardo. S. 2010. Evaluation of the biogas productivity potential of some Italian agro-industrial biomasses. Bioresource Technology. 101, 3780–3783.

Dressler, D.; Loewen, A.; Nelles, M. 2012. Life cycle assessment of the supply and use of bioenergy: impact of regional factors on biogas production. The International Journal of Life Cycle Assessment. 17(9), 1104-1115.

Dwidar, M., Lee, S., and Mitchell, R. 2012. The production of biofuels from carbonated beverages. Appl. Energ. 100; 47–51

Dwyer, K., Hosseinian, F. and Rod, M., 2014. The Market Potential of Grape Waste Alternatives. Journal of Food Research, 3(2), p.p91.

Eastman, J. A., and Ferguson, J. F. 1981. Solubilization of particulate organic carbon during the acid phase of anaerobic digestion. Journal (Water Pollution Control Federation), 352-366.

Ebner, J., Babbitt, C., Winer, M., Hilton, B., & Williamson, A. 2014a. Life cycle greenhouse gas (GHG) impacts of a novel process for converting food waste to ethanol and co-products. Applied Energy, 130, 86-93.

Ebner, J., Win, S.S., Hegde, S., Vadney, S., Williamson, A. and Trabold, T., 2014b, June. Estimating the biogas potential from colleges and universities. In ASME 2014 8th International Conference on Energy Sustainability collocated with the ASME 2014 12th International Conference on Fuel Cell Science, Engineering and Technology (pp. V002T04A005-V002T04A005). American Society of Mechanical Engineers.

Ebner, J. and Toaszewski, B., From Bane to Benefit: Mapping more effective food waste utilization. 2014c, ArcUser, ESRI, Spring 2014b

Ebner, J., Williamson, A., Trabold, T. and Tomaszewski, B., Organic Waste Resource Locator. 2014d, http://www.rit.edu/affiliate/nysp2i/organic-resource-locator-0

Ebner, J.; Williamson, A.; Trabold, T. Quantifying the greenhouse gas impact of pathways for treatment of secondary resources generated in the food supply chain. Proceedings of the ASME 2015a, Power and Energy Conversion Conference, PowerEnergy2015. June 28-July 2, 2015, San Diego, California.

Ebner, J.H., Labatut, R.A., Rankin, M.J., Pronto, J.L., Gooch, C.A., Williamson, A.A. and Trabold, T.A., 2015b. Lifecycle Greenhouse Gas Analysis of an Anaerobic Codigestion Facility Processing Dairy Manure and Industrial Food Waste. Environmental science & technology, 49(18), pp.11199-11208.

Ebner, J., Labatut, R., Lodge, J., Williamson, A., Trabold, T., 2016, Anaerobic co-digestion of commercial food waste and dairy manure: characterizing biochemical parameters and synergistic effects, Waste Management.

EcoInvent Centre. EcoInvent data v.2.2. EcoInvnent Reports No. 1-250. 2007, Swiss Centre for Life Cycle Inventories. Dubendorf. Available from www.ecoinvent.org

Erdei B., Barta Z., Sipos B., Réczey K., Galbe M., and Zacchi G. 2010. Ethanol production from mixtures of wheat straw and wheat meal. Biotechnol. Biofuels 3; 16.

Elbadrawy, E., & Sello, A. 2011. Evaluation of nutritional value and antioxidant activity of tomato peel extracts. Arabian Journal of Chemistry.

El-Mashad, H.M., Zhang, R., 2010. Biogas production from co-digestion of dairy manure and food waste. Bioresource Technology, 101, 4021-4028.

Esposito, G., Frunzo, L., Giordano, A., Liotta, F., Panico, A., & Pirozzi, F. 2012. Anaerobic co-digestion of organic wastes. Reviews in Environmental Science and Bio/Technology, 11(4), 325-341.

Esselen, W.B. and Fellers, C.R., 1939. The nutritive value of dried tomato pomace. Poultry Science, 18(1), pp.45-47.

Farrell, A., Plevin, R., Turner, B., Jones, A., O'Hare, M., and Kammen, D. 2006. Ethanol can contribute to energy and environmental goals. Science. 311(5760); 506–508.

Farrell, A., Plevin, R., Turner, B., Jones, A., O;Hare, M. and Kammen, D. Supporting online material for Ethanol Can Contribute to Energy and Environmental Goals. Renewable and Appropriate Energy Laboratory, Bekeley http://rael.berkeley.edu/sites/default/files/EBAMM/

Fish, W., Bruton, D., and Russo, V. 2009. Watermelon juice: a promising feedstock supplement, diluents and nitrogen supplement for ethanol biofuel production. Biotechnol. Biofuels. 2; 18.

Food and Agricultural Organization (FAO) of the United Nations, FAO's program on prevention of losses. E. De Las Casas, Plant Production and Protection Division, FAO, Rome, Italy, 1976, http://spiru.cgahr.ksu.edu/proj/iwcspp/pdf2/3/621.pdf

Food and Agriculture Organization (FAO) of the UN, “Launch of the G20 Technical Platform on the Measurement and Reduction of Food Loss and Waste”, May, 2015, http://www.fao.org/about/meetings/council/cl153/side-events/technical-platform/en/

Flesch, T., Desjardins, R.,2009. Worth, D. Fugitive methane emissions from an agricultural biodigester. Biomass and Bioenergy. 2011, 35 , 3927- 3935.

Frear, C, Zhao, B, Fu, G, Richardson, M, Chen, C and Fuchs, M. Biomass Inventory and Bioenergy Assessment: An Evaluation of Organic Material Resources for Bioenergy Production in Washington State, Publication No. 05-07-04, December 2005.

Fruergaard, T., Ekvall, T., & Astrup, T. 2009. Energy use and recovery in waste management and implications for accounting of greenhouse gases and global warming contributions. Waste Management & Research.

Friðriksson GB, Johnsen T, Bjarnadóttir HJ. 2002. Guidelines for the use of LCA in the waste management sector. (1537).

Griffin, M., Sobal, J., & Lyson, T. A. 2009. An analysis of a community food waste stream. Agriculture and Human Values, 26(1-2), 67-81

International S. Organization. ISO 14040:2006 - Environmental management - Life cycle assessment - Principles and framework. 2006

Ghaly, A.E.; Mahmoud, N.S.; Rushton, D.G.; Arab, F. 2007. Potential environmental and health impacts of high land application of cheese whey. Am. J. Agric. Biol. Sci. 2, 106–117.

Ghaly, A. E., & El-Taweel, A. A. 1997. Kinetic modelling of continuous production of ethanol from cheese whey. Biomass and Bioenergy, 12(6), 461-472.

Gentil, E. C., Damgaard, A., Hauschild, M., Finnveden, G., Eriksson, O., Thorneloe, S., ... & Christensen, T. H., 2010. Models for waste life cycle assessment: review of technical assumptions. Waste Management, 30(12), 2636-2648.

Gioelli, F.; Dinuccio, E. and Balsari, P. 2011. Residual biogas potential from the storage tanks of non-separated digestate and digested liquid fraction. Bioresource Technology. 102, 10248-10251.

Gooch, C. and Labatut, R. Anaerobic Digestion Monitoring at Synergy Biogas LLC; Cornell University Agreement No. 65961. Evaluation of the Continuously-Mixed Anaerobic Digester System at Synergy Biogas following the ERG Protocol (Jun 2012 to May 2014). www.manuremanagement.cornell.edu

Gooch, C.A.; Pronto, J.; Labatut, R. New York State Energy Research and Development Authority (NYSERDA), Biogas Distributed Generation Systems Evaluation and Technology Transfer, NYSERDA Project No. 6597; Evaluation of seven on-farm anaerobic digestion systems based on the ASERTTI monitoring protocol: Consolidated report and findings, PRO-DAIRY Program, Department of Biological and Environmental Engineering, Cornell University, Ithaca, NY, 2011. http://nmsp.cals.cornell.edu/publications/extension/Ndoc2003.pdf

Graboski, M. Fossil energy use in the manufacture of corn ethanol. National Corn Growers Association, Washington, DC; 2002.

Gunaseelan, V. N. 2004. Biochemical methane potential of fruits and vegetable solid waste feedstocks. Biomass and Bioenergy, 26(4), 389-399.

Gunders, Dana. Wasted: How America is losing up to 40 percent of its food from farm to fork to landfill. 2012. Natural Resources Defense Council Issue Paper. August, 2012. 26 pp

Gustafson, G.M., 2000. Partitioning of nutrient and trace elements in feed among milk, feces and urine by lactating dairy cows. Acta Agric. Scand. Sect. A – Anim. Sci. 50, 111–120.

Gustavsson, J., Cederberg, C., Sonesson, U., Van Otterdijk, R., & Meybeck, A. 2011. Global food losses and food waste. Food and Agriculture Organization of the United Nations, Rom.

Hansen, T.L.; Sommer, S.G.; Gabriel, S.; Christensen, T.H. Methane production during storage of anaerobically digested municipal organic waste. J. Environ. Qual. 2006, 35, 830–836.

Hansen, T. L., Schmidt, J. E., Angelidaki, I., Marca, E., Jansen, J. C., Mosbæk, H. & Christensen, T. H., 2004. Method for determination of methane potentials of solid organic waste Measurement of methane potentials of solid organic waste. Waste Management, 24(4), 393–400.

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.

Hoffmann, R.A., M.L. Garcia, M. Veskivar, K. Karim, M.H. Al-Dahhan, L.T. Angenent, 2008. Effect of shear on performance and microbial ecology of continuously stirred anaerobic digesters treating animal manure. Biotechnology and Bioengineering, 100, 38.
Hong. Y., and Yoon, H. Ethanol production from food residues. Biomass Bioenerg. 2011; 35(7); 3271–3275.

Hollingsworth, M. D., Shanklin, C. W., Gench, B., & Hinson, M. (1992). Composition of waste generated in six selected school food service operations. Food Service Research Review, 16(2), 125-131.

Hollingsworth, M. D., Shanklin, C. W., & Cross, E. W. (1995). Waste stream analyses in seven selected school food service operations. School Food Service Research Review, 19, 81-87

Hongladarom, Soraj, ed. Food Security and Food Safety for the Twenty-first Century: Proceedings of APSAFE2013. Springer, 2015.

Huijsmans, J.; Hol, J.; Vermeulen, G. 2003. Effect of application method, manure characteristics, weather and field conditions on ammonia volatilization from manure applied to arable land. Atmospheric Environment. 37(26), 3669-3680.

Infogroup, Inc. 2014. Hershey Company. Retrieved November 14, 2014, from ReferenceUSA database.

International Panel on Climate Change . IPCC, 1997. Guidelines for National Greenhouse Gas Inventories: Reference Manual. IPCC Guidelines, 1997

Intergovernmental Panel on Climate Change (IPCC), IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan 2006. http://www.ipcc-nggip.iges.or.jp/public/2006gl/

International Panel on Climate Change. IPCC Fourth Assessment Report: Climate Change 2007 (AR4). [cited 2012 December 8] Available from:

http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml#.UOdUmXe1Vv8

Intergovernmental Panel on Climate Change (IPCC). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp, doi:10.1017/CBO9781107415324.

Intergovernmental Panel on Climate Change (IPCC), Published: IGES, Japan http://www.ipcc-nggip.iges.or.jp/public/2006gl/

International Standards Organization. ISO 14040:2006 - Environmental management - Life cycle assessment - Principles and framework. 2006. International S. Organization

Intergovernmental Panel on Climate Change (IPCC), 2013: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp, doi:10.1017/CBO9781107415324

Jensen, P. D., Ge, H., and Batstone, D. J. 2011. Assessing the role of biochemical methane potential tests in determining anaerobic degradability rate and extent. Water Science and Technology, 64(4), 880.
Kalogo, Y., Habibi, S., MacLean, H., and Joshi, S. Environmental implications of municipal solid waste-derived ethanol. Environ. Sci. Technol. 2007; 41(1); 35–41.

Kantor, L.S., Lipton, K., Manchester, A., and Oliveira, V., "Estimating and Addressing America's Food Losses" USDA, Food Review 20.1 (1997): 2-12.

Khalid, A., Arshad, M., Anjum, M., Mahmood, T., and Dawson, L. 2011. The anaerobic digestion of solid organic waste. Waste Management, 31(8), 1737-1744.

Ketterings, Q.; Klausner, S., Czymmek, K. Nitrogen guidelines for field crops in New York. Second Release. June 22, 2003. Cornell University Department of Crop and Soil Sciences Extension Series E03-16. 2003.

Kim J., Oh B., Shin H.-J., Eom C.-Y., and Kim S. 2008. Statistical optimization of enzymatic saccharification and ethanol fermentation using food waste. Process Biochem. 43(11); 1308–1312.

Kim, Mi-Hyung, and Jung-Wk Kim. 2010. Comparison through a LCA evaluation analysis of food waste disposal options from the perspective of global warming and resource recovery. Science of the total environment 408.19: 3998-4006.

Kirkendall, N. J. (Ed.). 2015. Data and Research to Improve the US Food Availability System and Estimates of Food Loss: A Workshop Summary. National Academies Press.

Kumar, S., Singh, S., Mishra, I., and Adhikari, D. 2009. Recent advances in production of bioethanol from lignocellulosic biomass. Chem. Eng. Technol. 32(4); 517–526.

Kushwaha J.; Srivastava, V.C.; Mall, I.D. 2011. An overview of various technologies for the treatment of dairy wastewaters. Critical Reviews in Food Science and Nutrition. 51, 442–452.

Labatut, R. A.; Angenent, L. T.; Scott, N. R. 2011. Biochemical methane potential and biodegradability of complex organic substrates. Bioresource Technology. 102(3), 2255-2264.

Lane, A. G., 1983. Anaerobic digestion of spent coffee grounds. Biomass 3(4), 247-268.

Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., ... & Christensen, T. H. 2014. Review of LCA studies of solid waste management systems–Part I: Lessons learned and perspectives. Waste management, 34(3), 573-588.

Levis, J. M. and Barlaz, M.A. 2011. Is biodegradability a desirable attribute for discarded solid waste? Perspectives from a national landfill greenhouse gas inventory model. Environ. Sci. and Tech. 45 (13), 5470 – 76.

Levis, J. W., & Barlaz, M. A. 2011. What is the most environmentally beneficial way to treat commercial food waste?. Environmental science & technology,45(17), 7438-744.

Lehtomäki, A.; Huttunen, S.; Lehtinen, T.; Rintala, J. 2008. Anaerobic digestion of grass silage in batch leach bed processes for methane production. Bioresource Technology. 99, 3267–3278.

Liebetrau, J.; Reinelt, T.; Clemens, J. Hafermann, C.; Friehe, J.; Weilandrom, P. 2013. Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector. Water Science & Technology. 67 (6), 1370-1379.

Lipinski, B., Hanson, C., Lomax, J., Kitinoja, L., Waite, R., & Searchinger, T. 2013. Reducing food loss and waste. World Resources Institute Working Paper, June.

Luste, S. and Luostarinen, S. 2010. Anaerobic co-digestion of meat-processing by-products and sewage sludge –Effect of hygienization and organic loading rate. Bioresource Technology. 101, 2657–2664.

Lundqvist, J., de Fraiture, C., & Molden, D. 2008. Saving water: from field to fork: curbing losses and wastage in the food chain.

Ma, J., 2006. A web-based spatial decision support system for utilizing organic wastes as renewable energy resources in New York State., Cornell Disertation Thesis, 2006

Ma, H., Wang, Q., Gong, L., Wang, X. Wang, X., and Yin, W. 2008. Ethanol production from kitchen garbage by Zymomonas mobilis: Optimization of parameters through statistical experimental designs. Chem. Biochem. Eng. Q. 22(3); 369–375.

Massachusets Department of Environmental Protection (MDEP), 2002. Identification, Characterization, and Mapping of Food Waste and Food Waste Generators in Massachusetts; FINAL REPORT, prepared by Draper/Lennon, September 19, 2002

"Massachusetts Formally Proposes Commercial Food Waste Ban", July 12, 2013, Biocycle, http://www.biocycle.net/2013/07/12/massachusetts-formally-proposes-commercial-food-waste-ban/

Mata-Alvarez, J., Dosta, J., Macé, S., & Astals, S. 2011. Codigestion of solid wastes: a review of its uses and perspectives including modeling. Critical reviews in Biotechnology, 31(2), 99-111.

Matteson, G. C., and Jenkins, B. M. 2007. Food and processing residues in California: Resource assessment and potential for power generation.Bioresource Technology, 98(16), 3098-3105.

Menardo, S.; Giolli, F.; Balsari, P. 2011. The methane yield of digestate: effect of organic loading rate, hydraulic retention time, and plant feeding. Bioresource Technology. 102, 2348-51.

Meyer-Aurich, A.;,Schattauer, A.,Hellebrand, H.J.,Klauss, H;,Plöchl, M. 2012. Berg, W. Impact of uncertainties on greenhouse gas mitigation potential of biogas production from agricultural resources. Renewable Energy. 37, 277-284.

Møller, J.; Boldrin, A.; Christensen, T.H. 2009. Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution. Waste Manage Res., 27, 813–824.

Menardo, S., & Balsari, P., 2012. An analysis of the energy potential of anaerobic digestion of agricultural by-products and organic waste. BioEnergy Research, 5(3), 759-767.

Mussatto, S. I., Dragone, G., & Roberto, I. C. 2006. Brewers' spent grain: generation, characteristics and potential applications. Journal of Cereal Science, 43(1), 1-14.

National Research Council. Nutrient Requirements of Beef Cattle: Seventh Revised Edition: Update 2000. Washington, DC: The National Academies Press, 2000.
NDSU Extension. By-Products and Regionally Available Alternative Feedstuffs for Dairy Cattle J. W. Schroeder, 2012

Neves, L., Oliveira, R., & Alves, M. M. 2006. Anaerobic co-digestion of coffee waste and sewage sludge. Waste Management, 26(2), 176-181.

Nielfa, A., Cano, R., & Fdz-Polanco, M. 2015. Theoretical methane production generated by the co-digestion of organic fraction municipal solid waste and biological sludge. Biotechnology Reports, 5, 14-21.

Nielsen, O.K, Lyck, E., Mikkelsen, M.H., Hoffmann, L., Gyldenkærne, S., Winther, M., Nielsen, M., Fauser, P., Thomsen, M.,Plejdrup, M.S.; Illerup, J.B.; Sørensen, P.B.; Vesterdal, L. Denmark's national inventory report 2008 – Emission inventories 1990–2006. 2008, Submitted under the United Nations Framework Convention on Climate Change. NERI Technical Report no. 667. National Environmental Research Institute, University of Aarhus, Denmark. http://www.dmu.dk/Pub/FR667.pdf

Nigam, J. 2000. Continuous ethanol production from pineapple cannery waste using immobilized yeast cells. J. Biotechnol.; 80(2); 189–193.

North Carolina Department of Environment and Natural Resources (NCDENR), 2012. North Carolina 2012 Food Waste Generation, August 2012

NY Military bases, http://militarybases.com/new-york/ accessed June 2014

The Military Zone, http://themilitaryzone.com/bases/ accessed June 2014

Office of the State Comptroller (OSC), The Importance of Agriculture to the New York State Economy, Thomas P. DiNapoli New York State Comptroller, March 2015, http://www.osc.state.ny.us/reports/importance_agriculture_ny.pdf

Olofsson, K., Bertilsson, M., and Lidén, G. 2008. A short review on SSF — an interesting process option for ethanol production from lignocellulosic feedstocks. Biotechnol. Biofuels 2008; 1; 7.

Östergren, K., J. Gustavsson, H. Bos-Brouwers, T. Timmermans, O. J. Hansen, H. Møller, G. Anderson et al. 2014. FUSIONS Definitional Framework for Food Waste. EU FUSIONS, 2014.

Owen, W., D. Stuckey, J. Healy Jr., Young, and P. McCarty, 1979. Bioassay for monitoring biochemical methane potential and anaerobic toxicity. Water Research, 13:485-492.

Parfitt, Julian, Mark Barthel, and Sarah Macnaughton. 2010.Food waste within food supply chains: quantification and potential for change to 2050. Philosophical Transactions of the Royal Society B: Biological Sciences 365.1554 : 3065-3081

Peletz, S, personal communication with Steve Peletz, Monroe County Department of Environmental Services, March 3, 2014

Pronto, J. L., and Gooch, C. A. Greenhouse gas emission reductions for seven on-farm dairy manure-based anaerobic digestion systems–final results. In ASABE–International Symposium on Air Quality Waste Management and Agriculture. 2010, Vol. 7, pp. 35-42.

Poeschl, M.; Ward, S.; Owende, P. 2012. Environmental impacts of biogas deployment–Part I: Life cycle inventory for evaluation of production process emissions to air. Journal of Cleaner Production. 24, 168-183.

Poeschl, M., Ward, S.,Owende, P. 2012. Environmental impacts of biogas deployment–Part II: life cycle assessment of multiple production and utilization pathways. Journal of Cleaner Production, 24, 184-201.

Qiao, W., Takayanagi, K., Shofie, M., Niu, Q., Yu, H. Q., & Li, Y. Y. 2013. Thermophilic anaerobic digestion of coffee grounds with and without waste activated sludge as co-substrate using a submerged AnMBR: system amendments and membrane performance. Bioresource Technology, 150, 249-258.

Raposo, F., Fernández-Cegrí, V., De la Rubia, M. A., Borja, R., Béline, F., Cavinato, C., ... & De Wilde, V. 2011. Biochemical methane potential (BMP) of solid organic substrates: evaluation of anaerobic biodegradability using data from an international interlaboratory study. Journal of Chemical Technology and Biotechnology, 86(8), 1088-1098.

Reichert, P. 2014. AQUASIM a tool for simulation and data analysis of aquatic systems. Water Science & Technology, 30(2), 21-30.

Razaviarani, V.; Buchanan, I.D.; Malik, S.; Katalambula, H. 2013. Pilot-scale anaerobic co-digestion of municipal wastewater sludge with restaurant grease trap waste. Journal of Environmental Management.123, 26-33.

Risse, L.M.; Cabrera, M.L.; Franzluebbers, A.J.; Gaskin, J.W.; Gilley, J.E.; Killorn, R.; Radcliffe, D.E.; Tollner, E.W.; Zhang, H. Land application of manure for beneficial reuse. 2001, In White Papers on Animal Agriculture and the Environment; National Center for Manure and Animal Waste Management: Raleigh, N.C., 7.1–7.46.

Rieger, C. and Weiland, P. Biogas – Messprogramm I 61 Biogasanlagen im Verlich. Erstellt durch Johann Heinrich vonThunen-Institut (VTI), 2009, Gulzow, Germany.

Rico, C., Muñoz, N., Fernández, J., & Rico, J. L. 2015. High-load anaerobic co-digestion of cheese whey and liquid fraction of dairy manure in a one-stage UASB process: Limits in co-substrates ratio and organic loading rate. Chemical Engineering Journal, 262, 794-802.

Rodriguez-Verde, I., Regueiro, L.,Carballa; M., Hospido A., Lema, J. M., 2014. Assessing anaerobic co-digestion of pig manure with agroindustrial wastes: The link between environmental impacts and operational parameters. Science of the Total Environment. 497, 475-483.

Safferman, S., Wright, T., Miller, S., & Higginbotham, A. 2007. Wastewater Characteristics and Quantities Associated with Fruit and Vegetable Processing in Michigan. Michigan State University: East Lansing, MI, USA.

Shalini, R. and Gupta, D.K., 2010. Utilization of pomace from apple processing industries: a review. *Journal of food science and technology*,*47*(4), pp.365-371.

Schmitt, E., Bura, R., Gustafson, R., Cooper, J., and Vajzovic, A. Converting lignocellulosic solid waste into ethanol for the State of Washington: An investigation of treatment technologies and environmental impacts. Biores. Technol. 2012, 104; 400–409.

Soethoudt, H., and Timmermans, T. 2013. Food waste monitor: Mid--term report.

Staley, B. F. and Barlaz, M. A. 2009. Composition of municipal solid waste in the United States and implications for carbon sequestration and methane yield. Journal of Environmental Engineering. 135 (10), 901-909.

Smith, A., Brown, K., Ogilvie, S., Rushton, K., & Bates, J. 2001. Waste management options and climate change. Final Report ED21158R4, 1, 205.

Stampe, S., Alcock, R., Westby, C. and Chisholm, T. 1983. Energy consumption of a farm-scale ethanol distillation system. Energ. Agr. 2; 355–368.

Stehfest, E. and Bouwman, L. 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutrient Cycling in Agroecosystems. 74, 207 –228.

Tagtow, A., Nguyen, J., Johnson-Bailey, D., & Schap, T. E. 2015. Food Waste Reduction Efforts at the USDA. Journal of the Academy of Nutrition and Dietetics, 115(12), 1914-1918.

Tang Y., Zhao D., Cristhian C., and Jiang, J. 2011. Simultaneous saccharification and cofermentation of lignocellulosic residues from commercial furfural production and corn kernels using different nutrient media. Biotechnol. Biofuels. 4; 22.

UN News center, “Transforming our world: the 2030 Agenda for Sustainable Development”, United Nations Sustainable Development Summit 2015, 25 - 27 September 2015, New York https://sustainabledevelopment.un.org/post2015/transformingourworld

UAD, University of Alaska Fairbanks cooperative extenstion, Using spent brewery grain in the Alaska compost pile, publication HGA-01026, https://www.uaf.edu/files/ces/publications-db/catalog/anr/HGA-01026.pdf

U.S. Census Bureau. Economic Census. Washington, D.C. Available at http://www.census.gov/econ.

USDA National Agricultural Statistics Service (NASS), NYS Agricultural Overview 2014, http://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=NEW%20YORK

USDA, National Nutrient Database for Standard Reference, http://ndb.nal.usda.gov/, 2105a

USDA Economic Research Service (ERS), Background and history of food loss estimates, web access 2015b, http://www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system/loss-adjusted-food-availability-documentation.aspx

USDA/US EPA. Waste Not, Want Not. Feeding the Hungry and Reducing Solid Waste Through Food Recovery. EPA 530-R-99-040.

U.S. EPA, Emissions & Generation Resource Integrated Database (eGRID), 9th edition v1.0, year 2010 emissions. 2013. http://www.epa.gov/cleanenergy/energy-resources/egrid/index.html

U.S. EPA. Solid Waste Management and Greenhouse Gases; A Life-Cycle Assessment of Emissions and Sinks, 3rd Edition, 2006. [cited 2013 March 20] Available from: http://www.epa.gov/climatechange/wycd/waste/downloads/fullreport.pdf

U.S. EPA, Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2012. Environmental Protection Agency (EPA), 2014, http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2014-Main-Text.pdf

U.S. EPA. Municipal Solid Waste Generation, Recycling, and Disposal in the United States Tables and Figures for 2010. [cited 2012 Dec 10] Available from: http://www.epa.gov/osw/nonhaz/municipal/msw99.htm

U.S. EPA. “Waste” In: Inventory of US Greenhouse Gas Emissions and Sinks; 1990-2011, 2013. [cited 2013 January 10] Available from: http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2013-Chapter-8-Waste.pdf

U.S. EPA. Waste Reduction Model (WARM) v.12, 2012a. [cited 2013 January 10] Available from: http://www.epa.gov/wastes/conserve/tools/warm/Warm_Form.html

U.S. EPA. "Landfilling" In: WARM User's Guide v12 , 2012b. [cited 2013 January 10] Available from: http://epa.gov/epawaste/conserve/tools/warm/pdfs/Landfilling.pdf

U.S. EPA. "Composting" In: WARM User's Guide, 2012c . [cited 2013 January 10] Available from: http://epa.gov/epawaste/conserve/tools/warm/pdfs/Composting.pdf

U.S. EPA, Emissions & Generation Resource Integrated Database (eGRID), 9th edition v1.0, year 2010 emissions. Environmental Protection Agency (EPA), http://www.epa.gov/cleanenergy/energy-resources/egrid/index.html

U.S. EPA, Landfill gas emissions model (LandGEM) v.3.02, 2015b, http://www3.epa.gov/ttn/catc/products.html#software

U.S. EPA. Landfill Gas Emissions Model (LandGEM) version 3.02 user's guide. May 2005. [cited 2013 October 15] Available from: http://www.epa.gov/ttn/catc/dir1/landgem-v302-guide.pdf

U.S. EPA, LFG Energy Project Development Handbook, Feb 2015c, http://www3.epa.gov/lmop/documents/pdfs/pdf_full.pdf

U.S. EPA, Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM) March 2015d

U.S. EPA, Renewable Portfolio Standards (RPS), web, 2015, http://www3.epa.gov/epawaste/hazard/wastemin/minimize/energyrec/rps.htm

U.S. Life Cycle Inventory (USLCI) Database. 2012, National Renewable Energy Laboratory, 2012: https://www.lcacommons.gov/nrel/search

UNFCCC/CCNUCC, CDM Methodological to determine methane eemissions avoided from disposal of waste at a solid waste disposal site. 2008, Report Annex10 v. 4. August 2008. [Retrieved January 15, 2014] http://cdm.unfccc.int/EB/026/eb26_repan14.pdf

Velthof, C.; van Bruggen, C.M.; Groenestein, de Haan, B.J.; Hoogeveen, M.W.; Huijsmans, J.F.M. A model for inventory of ammonia emissions from agriculture in the Netherlands. Atmospheric Environment. 2011, 46,248-255.

Wagner, Steven, VP, Merrick and Company, personal communication, March 2013.

Wang, M. Updated energy and greenhouse gas emission results of fuel ethanol, 15th Intl. Symp. on Alcohol Fuels, 26–28 September 2005, San Diego, CA [cited 2012 March 20] Available from: http://www.eri.ucr.edu/ISAFXVCD/ISAFXVPP/UGEEERF.pdf

Wang M. GREET1_12 —Transportation Fuel Cycles Model. Argonne National Laboratory, Argonne, IL; 2012 [cited 2012 March 20] Available from: http://greet.es.anl.gov/publications

Watkins, M. and Nash, D. Factory wastewaters, their use on land and possible environmental impacts – A Mini Review. The Open Agriculture Journal. 2010, 4, 1-9.

Weidema, B.P.; Bauer, Ch.; Hischier, R.; Mutel, Ch.; Nemecek, T.; Reinhard, J.; Vadenbo, C.O.; Wernet, G, 2013, The ecoinvent database: Overview and methodology, Data quality guideline for the ecoinvent database version 3, www.ecoinvent.org

Wilkie, A.C., Graunke, R.E. and Cornejo, C., 2015. Food Waste Auditing at Three Florida Schools. Sustainability, 7(2), pp.1370-1387.

Wood, S. and Cowie, A. A review of greenhouse gas emission factors for fertilizer 2004, Research and Development Division, State Forests of New South Wales. Cooperative Research Centre for Greenhouse Accounting, For IEA Bioenergy Task 38.

Woon, S. and Othman, M. Anaerobic Digestion of Meat Wastes. 2012, 2012 International Conference on Clean and Green Energy IPCBEE vol.27, IACSIT Press, Singapore.

Yoshida, H., Nielsen, M. P., Scheutz, C., Jensen, L. S., Bruun, S., & Christensen, T. H. (2015). Long-Term Emission Factors for Land Application of Treated Organic Municipal Waste. Environmental Modeling & Assessment, 1-14.