EVALUATION OF FOOD WASTE DIVERSION POTENTIAL AND ECONOMICS OF USING FOOD WASTE DEHYDRATORS

by

ARINDAM DHAR

Presented to the Faculty of the Graduate School of
The University of Texas at Arlington in Partial Fulfillment
of the Requirements
for the Degree of

MASTER OF SCIENCE IN CIVIL ENGINEERING

THE UNIVERSITY OF TEXAS AT ARLINGTON
July 2016

Acknowledgments

First and foremost, I would like to extend my gratitude and thankfulness to my academic advisor and thesis supervisor, Dr. Sahadat Hossain, for his motivation, support and invaluable inputs during my entire graduate studies. Without his constant supervision, guidance and unflinching support, this thesis would not have been completed.

I would also like to appreciate the Dr. Xinbao Yu and Dr. Laureano Hoyos for their consent to serve in my committee and provide their valuable time and suggestions. I sincerely acknowledge the City of Denton landfill for their co-operation and assistance in sample collection and food waste dehydrator operation.

I am indebted to Dr. Sadik Khan, Dr. Sonia Samir and Md. Zahangir Alam for their invaluable support, critical inputs and guidance during my graduate studies. I would also like to gratefully acknowledge the contributions and assistance of Ishtiaque, Basit, Anuja, Rakib and Jobair during sample collection, sorting and testing, when I needed it the most.

Last but by no means the least, I sincerely convey my deepest gratitude to my parents, my family and my wife, Mandakini, for their constant motivation, support and for being an inspiration to all my success.

July 17, 2016.

Abstract

EVALUATION OF FOOD WASTE DIVERSION POTENTIAL AND ECONOMICS OF USING FOOD WASTE DEHYDRATORS

Arindam Dhar, MS

The University of Texas at Arlington, 2016

Supervising Professor: Dr. Sahadat Hossain

According to US EPA (2013), food waste is the second largest component (14.6%) of MSW composition in the US, preceded only by paper. Apart from the negative environmental impacts of food waste through methane and leachate generation, it also incurs huge cost to landfill them. Hence, food waste diversion from landfills is becoming a major issue in landfills across many US states. Though biological treatment of food waste is a potential alternative for food waste diversion, the existing scale of this type of treatment is very narrow. On-site treatment of food waste is a suitable alternative as it can be adopted across any scale. Among the on-site food waste treatment technologies, dehydrators present a sustainable benefit due to the retention of nutrient values in its output. However, no systematic studies has been undertaken to identify the diversion potential of food waste dehydrators and sustainable usage of its end products. Therefore, the objective of this study is to evaluate the potential of dehydrators to divert food waste from the landfill and propose sustainable usage of its end products, dry food waste and condensate water.

In the current study, a total of 8 samples were collected from 4 different sources to characterize the composition, moisture content and unit weight of the food waste samples. The food wastes were found to consist mostly of fruits and vegetables and grain products, with fruits and vegetable being the major component in most of the samples. The moisture content and the unit weight of the samples were found to vary between 63-87% and 1373-1828 lb/yd^3 respectively. The food waste dehydrator achieved a weight loss of 53-91% depending on sample sources. The dehydrator operation was found to be economically feasible for weight loss percentage more than 53% in the study conditions. The study also revealed that for the case of City of Denton Landfill, even a 50% food waste diversion to dehydrator may lead to daily airspace gain of 9 yd^3, valued at $ 185. The dry food waste was tested for its unit weight, particle size distribution, pH and C/N ratio and was found to possess the qualities of an ideal compost feedstock. The condensate water was tested for its pH, turbidity, BOD_5, fecal coliform and enterococci to determine its usability as reclaimed water according to TCEQ regulations. Though the condensate water had acceptable quality parameters in all other categories, it was found to have exceedingly high BOD_5 (2200 – 9250 mg/l) and should be treated for BOD_5 removal before it can be used as reclaimed water. Also, the food waste dehydrator was found to be more efficient and economical than the other available on-site food waste processing alternatives.

TABLE OF CONTENTS

List of Figures

List of Tables

Chapter 1

Introduction

1.1 Background

According to Food and Agriculture Organization (FAO), food waste is the wholesome edible material intended for human consumption, arising at any point in the Food Supply Chain (FSC) that is instead discarded, lost, degraded or consumed by pests (FAO, 1981). Food waste continues to be the second largest component (14.6%) of MSW composition in the US (US EPA, 2013). 97% of generated food waste is landfilled, which incurs a huge cost ($1.2 billion) to dispose (Schwab 2010, cited in Buzby et al., 2011). Food waste negatively impacts the environment through methane generation and leachate production. Hence, food waste is a concern in all of the major economies in the world due to its environmental impacts and economic loss. Food waste diversion from landfills is becoming a major issue due to the progressively increasing organics ban in landfills in many US states.

According to Levis et al. (2010), biological treatment of food waste is a suitable alternative to achieve food waste diversion from the landfill. Aerobic Composting (AC) and Anaerobic Digestion (AD) are most commonly adopted techniques for off-site processing of food waste. Though AD is the most preferred option from environmental standpoint, only small scale digesters are presently operating in USA (capacity < 42,500 Mg/yr) (Levis et al., 2010). To cope up with the recent developments, Industrial-Commercial-Institutional (ICI) food waste generators are considering many technologies for on-site processing of food waste.

On-site food waste treatment technologies are broadly divided into two types: Biological digesters and non-biological systems. The biological digesters, both aerobic and anaerobic, has some limitations in their operations such as higher process control, addition of micro-organisms, selective waste processing, higher processing times and also incurs higher capital costs (Cook, 2015). However, the non-biological systems include pulpers/shredders and dehydrators, which are easier and more economical to operate, requires no external additives and has lower processing times. Among the on-site food waste treatment technologies, dehydrators present a sustainable benefit due to the retention of nutrient values in its output. The output of a typical dehydrator (250 pounds capacity) consists of 25 pounds of sterile organic biomass and 25 gallons of condensed water (Neale, 2013). However, due to relatively small usage of dehydrators in the US, wastewater authorities and regulatory officials have very limited knowledge of the systems and their output (BioCycle, 2013). The dehydrators, along with their potential sustainable end product usage, can achieve landfill volume saving and reduce environmental impact of food waste. This research focuses on estimating food waste diversion potential of dehydrators and their potential end product utilization.

1.2 Problem Statement

Progressive food waste disposal bans in many US states have led the ICI generators to adopt various on-site food waste processing systems. Among these systems, food waste dehydrators are non-biological, effective on-site treatment systems which achieve 70-90% volume and mass reduction (Spencer, 2008). The dehydrator output consists of sterile dehydrated organic biomass and condensate water. The suggested uses of the end products are use of the dehydrated organic biomass as soil amendments and condensate water for landscaping or other recoverable use.

However, some state legislation maintains that land application of food waste is disposal (CA Title 14, 6 CRR-NY). Moreover, a study by Bergstrom and Rasmussen (2011) found that the unprocessed dehydrated food waste samples were not suitable as a soil amendment and rehydration of the dehydrated food waste produces fungus. Hence, direct application of dehydrated food waste as soil amendment seems improbable. However, composters receiving output from dehydrators were satisfied with the feedstock due to its valuable nitrogen and carbon content (Neale, 2013). Additionally, the condensate water can be utilized for irrigation and landscape usage. However, no systematic studies has been undertaken to identify the diversion potential of dehydrators and sustainable usage of its end products. Hence, a comprehensive study to identify the diversion potential of the dehydrators, along with suitability of dehydrated biomass as compost feedstock and condensate water as reclaimed water is necessitated.

1.3 Research Objective

The main objective of this study is to investigate the suitability of using food waste dehydrator for possible food waste diversion from the landfill. Also, to evaluate the economic benefits of the waste diversion and suitability of using the end products from the food waste dehydrator, the following tasks will be undertaken in this study:

- Determination of the physical and chemical properties of the dehydrated food waste.
- Determination of the physical, chemical and biological properties of the condensate water from the dehydrator.
- Comparison of the obtained properties of dehydrated food waste and condensate water with compost feedstock and TCEQ regulations for reclaimed water respectively.
- Determination of economic benefit of food waste diversion by using food waste dehydrator.

1.4 Thesis Organization

This thesis consists of five chapters: Introduction (Chapter 1), Literature Review (Chapter 2), Methodology (Chapter 3), Results & Discussion (Chapter 4), Potential Benefits of Dehydrator Operation (Chapter 5) and Conclusions & Recommendation for future work (Chapter 6).

The first chapter introduces general information of the study, problem statement, research objectives and a brief outline of the thesis organization.

The second chapter highlights the literatures on food waste generation, composition of food waste, global and national extent of food waste, food waste management hierarchy and different off-site and on-site food waste processing systems.

The third chapter elaborates the process of sample collection, dehydrator operation, experimental setups, test matrix and necessary laboratory test methodologies to address the research objective.

The fourth chapter discusses the test results obtained from the tests performed on the dehydrator outputs (dehydrated organic biomass and condensate water) and estimates the diversion potential, potential usage of end products and economic feasibility of using dehydrator as an on-site food waste treatment system.

The fifth chapter illustrates the potential economic benefits of using dehydrators for on-site food waste processing, possible sustainable usage of its end products and compares it with the other available on-site waste processing alternatives.

The sixth chapter summarizes and presents the conclusions of the outlined study and provides recommendations for future research work.

Chapter 2

Literature Review

2.1 Introduction

According to Food and Agriculture Organization (FAO), food waste is the wholesome edible material intended for human consumption, arising at any point in the Food Supply Chain (FSC) that is instead discarded, lost, degraded or consumed by pests (FAO, 1981). Stuart (2009) suggests that food waste should also include edible material that is intentionally fed to animals or is a by-product of food processing diverted away from the human food chain. Stuart's (2009) definition of food waste provides a wider scope for food surplus and waste management opportunities, because it includes food losses due to animal feeding and the diversion of food processing by-products (Papargyropoulou et al., 2014).

Food losses take place at all stages (production, postharvest and processing) of the food supply chain (Parfitt et al., 2010). Food waste is a concern in all of the major economies in the world. Besides environmental impacts, food waste also imposes an economic cost on consumers and retailers. A proper estimation of food waste quantities could provide a unique incentive for emission mitigation and monetary saving through waste reduction (Venkat, 2011).

2.2 Global Extent of Food Waste

A study by the Food and Agriculture Organization (FAO) reported that one-third of all food produced for human consumption is lost or wasted globally, amounting to about 1.2 billion metric tons annually (FAO, 2011). Food is wasted throughout the FSC, starting from initial agricultural production to final household consumption (FAO, 2011). Edible food mass is lost, discarded or degraded in different stages of food supply chain like production, postharvest handling, processing, distribution and consumption (FAO, 2011; Parfitt et al., 2010; Galanakis, 2012). Figure 2.1 shows the activities that generates food waste in the FSC.

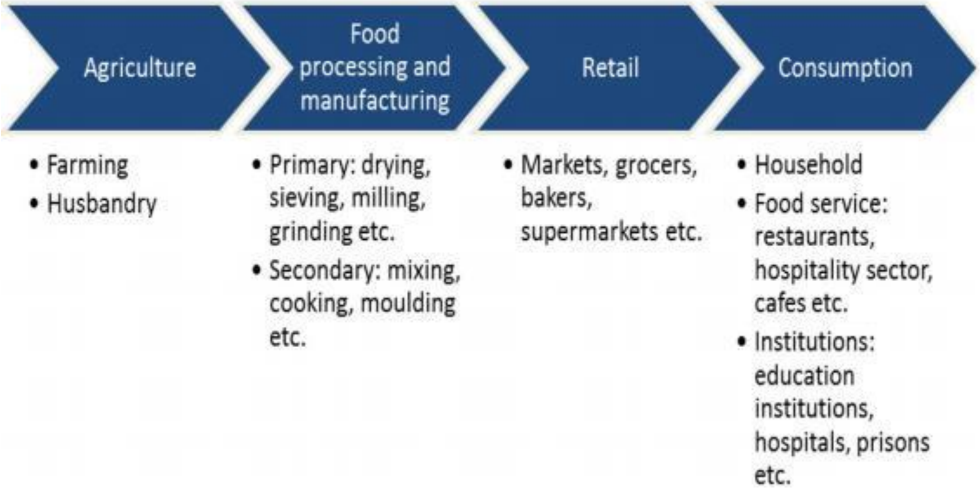


Figure 2.1 Activities giving rise to food losses and waste in the food supply chain (Papagyropoulou et al.,2014)

In medium and high-income countries, food is wasted to a great extent, both at consumer and production level. In contrast, in low-income countries, food is mainly lost during the early and middle stages of the food supply chain; much less food is wasted at the consumer level. Figure 2.2 illustrates the per capita food waste in different regions of the world. Per capita food wastage is highest in Europe and North-America (95-115 kg/year) and lowest in sub-Saharan Africa and South/Southeast Asia (6-11 kg/year) (FAO, 2011).

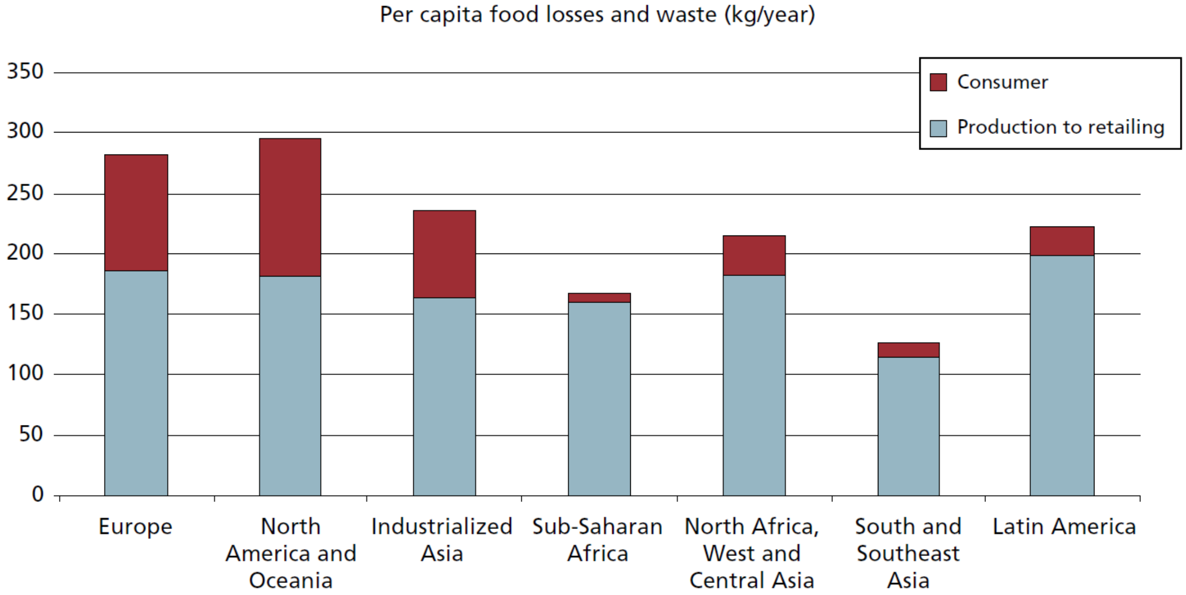


Figure 2.2 Per Capita food losses and waste in different regions of the world (FAO, 2011)

Food losses in industrialized countries are as high as in developing countries, but in developing countries more than 40% of the food losses occur at post-harvest and processing levels, while in industrialized countries, more than 40% of the food losses occur at retail and consumer levels. Food waste at consumer level in industrialized countries (222 million ton) is almost as high as the total net food production in sub-Saharan Africa (230 million ton) (FAO, 2011). Figure 2.3 and Table 1 shows the food waste production volume of each commodity group in different regions of the world.

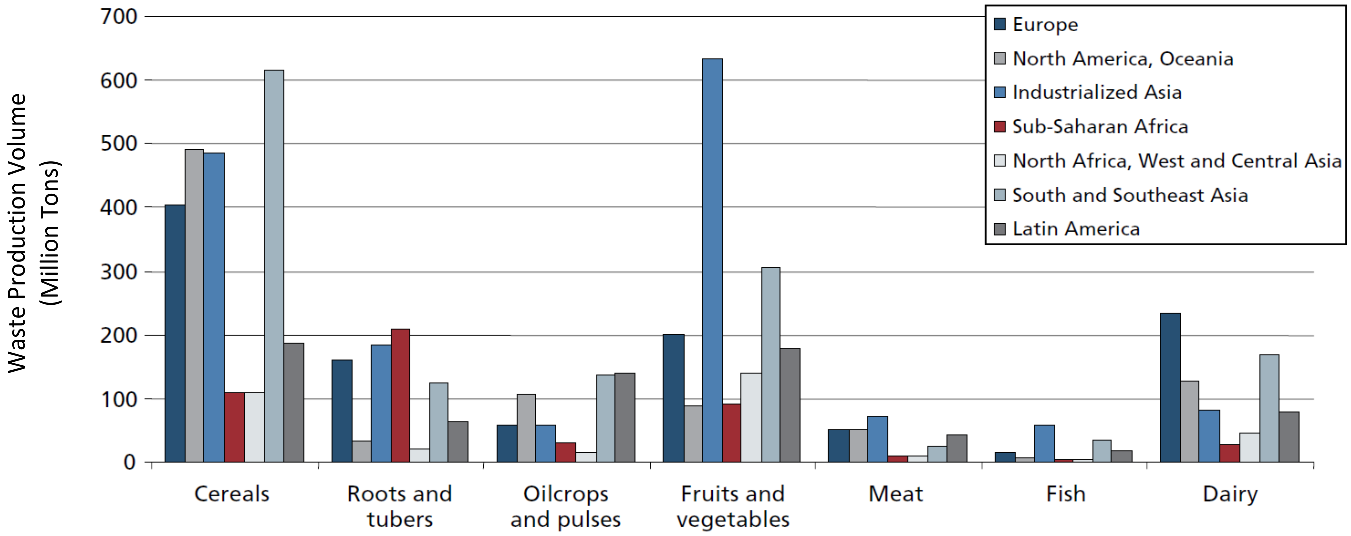
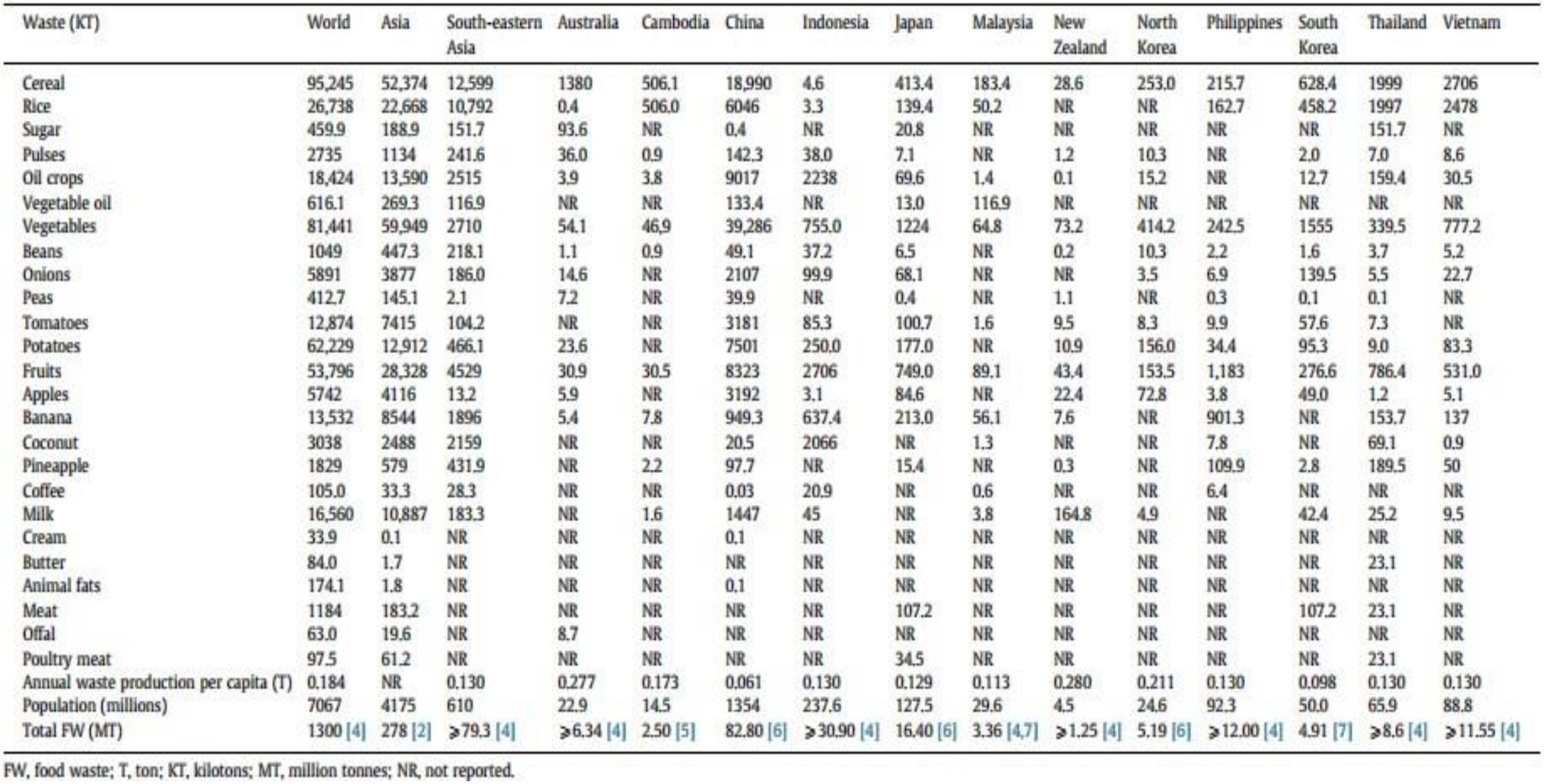


Figure 2.3 Food waste production volumes by commodity group in different regions (FAO, 2011)

Table 1 Typical food waste in Asia-Pacific countries and around the globe

(Kiran et al., 2014)



Waste (KT)	World	Asia	South-eastern Asia	Australia	Cambodia	China	Indonesia	Japan	Malaysia	New Zealand	North Korea	Philippines	South Korea	Thailand	Vietnam
Cereal	95,245	52,374	12,599	1380	506.1	18,990	4.6	413.4	183.4	28.6	253.0	215.7	628.4	1999	2706
Rice	26,738	22,668	10,792	0.4	506.0	6046	3.3	139.4	50.2	NR	NR	162.7	458.2	1997	2478
Sugar	459.9	188.9	151.7	93.6	NR	0.4	NR	20.8	NR	NR	NR	NR	NR	151.7	NR
Pulses	2735	1134	241.6	36.0	0.9	142.3	38.0	7.1	NR	1.2	10.3	NR	2.0	7.0	8.6
Oil crops	18,424	13,590	2515	3.9	3.8	9017	2238	69.6	1.4	0.1	15.2	NR	12.7	159.4	30.5
Vegetable oil	616.1	269.3	116.9	NR	NR	133.4	NR	13.0	116.9	NR	NR	NR	NR	NR	NR
Vegetables	81,441	59,949	2710	54.1	46,9	39,286	755.0	1224	64.8	73.2	414.2	242.5	1555	339.5	777.2
Beans	1049	447.3	218.1	1.1	0.9	49.1	37.2	6.5	NR	0.2	10.3	2.2	1.6	3.7	5.2
Onions	5891	3877	186.0	14.6	NR	2107	99.9	68.1	NR	NR	3.5	6.9	139.5	5.5	22.7
Peas	412.7	145.1	2.1	7.2	NR	39.9	NR	0.4	NR	1.1	NR	0.3	0.1	0.1	NR
Tomatoes	12,874	7415	104.2	NR	NR	3181	85.3	100.7	1.6	9.5	8.3	9.9	57.6	7.3	NR
Potatoes	62,229	12,912	466.1	23.6	NR	7501	250.0	177.0	NR	10.9	156.0	34.4	95.3	9.0	83.3
Fruits	53,796	28,328	4529	30.9	30.5	8323	2706	749.0	89.1	43.4	153.5	1,183	276.6	786.4	531.0
Apples	5742	4116	13.2	5.9	NR	3192	3.1	84.6	NR	22.4	72.8	3.8	49.0	1.2	5.1
Banana	13,532	8544	1896	5.4	7.8	949.3	637.4	213.0	56.1	7.6	NR	901.3	NR	153.7	137
Coconut	3038	2488	2159	NR	NR	20.5	2066	NR	1.3	NR	NR	7.8	NR	69.1	0.9
Pineapple	1829	579	431.9	NR	2.2	97.7	NR	15.4	NR	0.3	NR	109.9	2.8	189.5	50
Coffee	105.0	33.3	28.3	NR	NR	0.03	20.9	NR	0.6	NR	NR	6.4	NR	NR	NR
Milk	16,560	10,887	183.3	NR	1.6	1447	45	NR	3.8	164.8	4.9	NR	42.4	25.2	9.5
Cream	33.9	0.1	NR	NR	NR	0.1	NR	NR	NR	NR	NR	NR	NR	NR	NR
Butter	84.0	1.7	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	23.1	NR
Animal fats	174.1	1.8	NR	NR	NR	0.1	NR	NR	NR	NR	NR	NR	NR	NR	NR
Meat	1184	183.2	NR	NR	NR	NR	NR	107.2	NR	NR	NR	NR	107.2	23.1	NR
Offal	63.0	19.6	NR	8.7	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Poultry meat	97.5	61.2	NR	NR	NR	NR	NR	34.5	NR	NR	NR	NR	NR	23.1	NR
Annual waste production per capita (T)	0.184	NR	0.130	0.277	0.173	0.061	0.130	0.129	0.113	0.280	0.211	0.130	0.098	0.130	0.130
Population (millions)	7067	4175	610	22.9	14.5	1354	237.6	127.5	29.6	4.5	24.6	92.3	50.0	65.9	88.8
Total FW (MT)	1300 [4]	278 [2]	⩾79.3 [4]	⩾6.34 [4]	2.50 [5]	82.80 [6]	⩾30.90 [4]	16.40 [6]	3.36 [4,7]	⩾1.25 [4]	5.19 [6]	⩾12.00 [4]	4.91 [7]	⩾8.6 [4]	⩾11.55 [4]

FW, food waste; T, ton; KT, kilotons; MT, million tonnes; NR, not reported.

2.3 Food Waste in USA

Food production in USA uses 50% land, consumes 80% freshwater and accounts for 10% of the total U.S. energy budget (Ebel et al., 2011; Gunders, 2012; Webber, 2011). According to Buzby et al. (2012), 31 percent (133 billion pounds) of the available food supply at the retail and consumer levels in 2010 went uneaten. Retail-level and consumer level losses accounted for 10% and 21% of the available food supply respectively. The estimated total value of food loss at the retail and consumer levels in the United States was $161.6 billion in 2010. Meat, poultry, and fish (30 percent, $48 billion); vegetables (19 percent, $30 billion); and dairy products (17 percent, $27 billion) had the lion’s share of total value of food loss. The total amount of food loss represents 387 billion kcal per day in 2010. Recovery costs, food safety considerations, and other factors can reduce the amount of food recovered for human consumption (Buzby et al., 2012). Figure 2.4 represents the food loss at each steps in the supply chain in North America.

In 2013, Americans generated about 254 million tons of Municipal Solid Waste (MSW). Organic materials continue to be the largest component of MSW and food waste is the second largest component of the waste stream (US EPA, 2013). Figure 2.5 shows the MSW composition of USA in 2013. Less than 3% of food waste was recovered and recycled in 2008 (US EPA,2008) and more than 97% ended up in the landfill, which cost about $1.3 billion to landfill (Schwab 2010, cited in Buzby et al., 2011). These disposal methods has negative impacts on the environment due to methane generation, which is a by- product of anaerobic decomposition in the landfill. Additionally, produced leachate may contaminate groundwater if the landfills are not properly maintained (Buzby et al., 2011). UNEP emphasizes on the economic benefits of resource efficiency and waste reduction. UNEP suggests that minimization of resource use, waste and other emissions have the potential to yield cost savings, identify new business fields, and increase employment and competitiveness (UNEP, 2009).

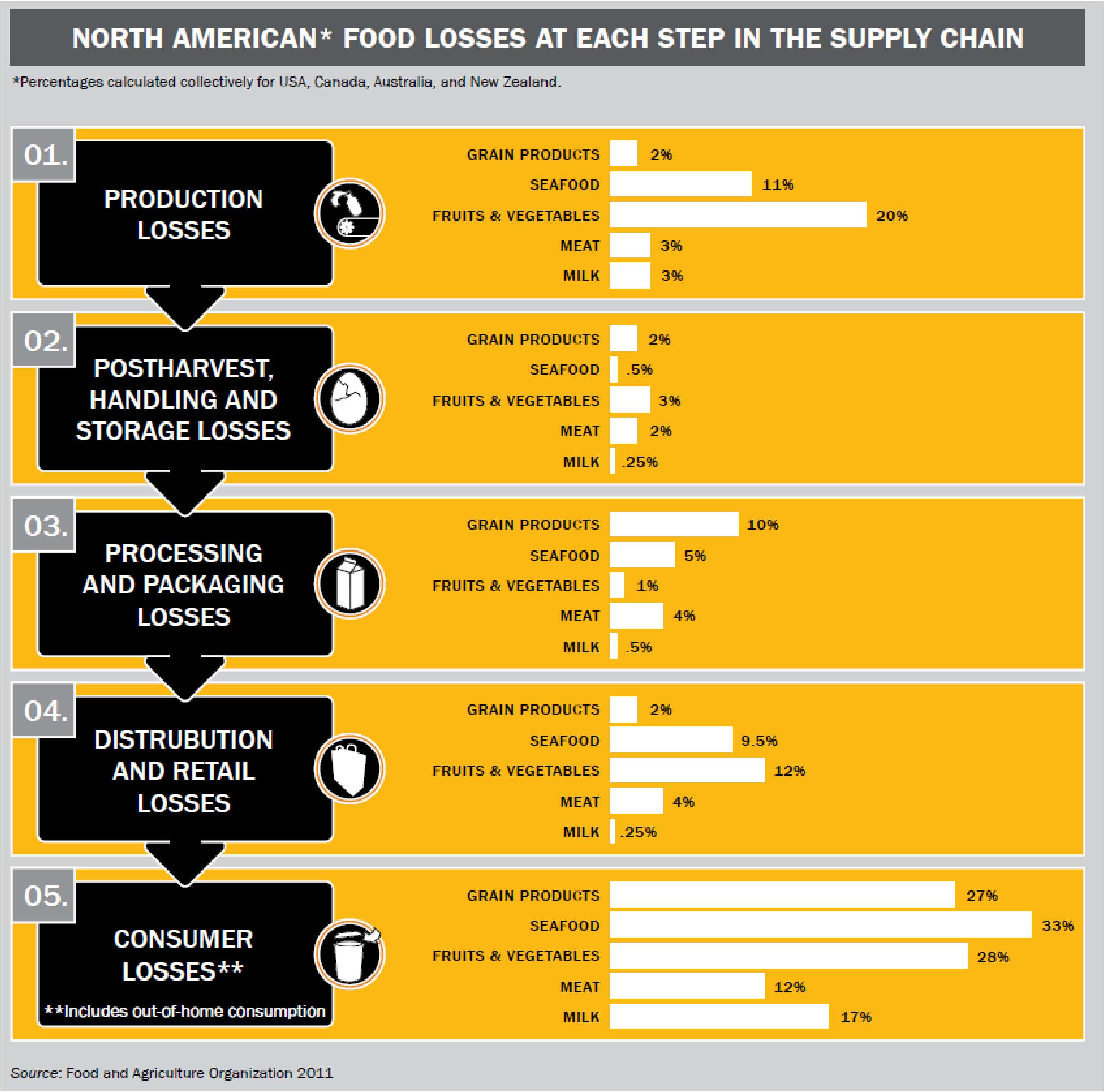


Figure 2.4 Food Losses in the Supply Chain in North America (Gunders, 2012)

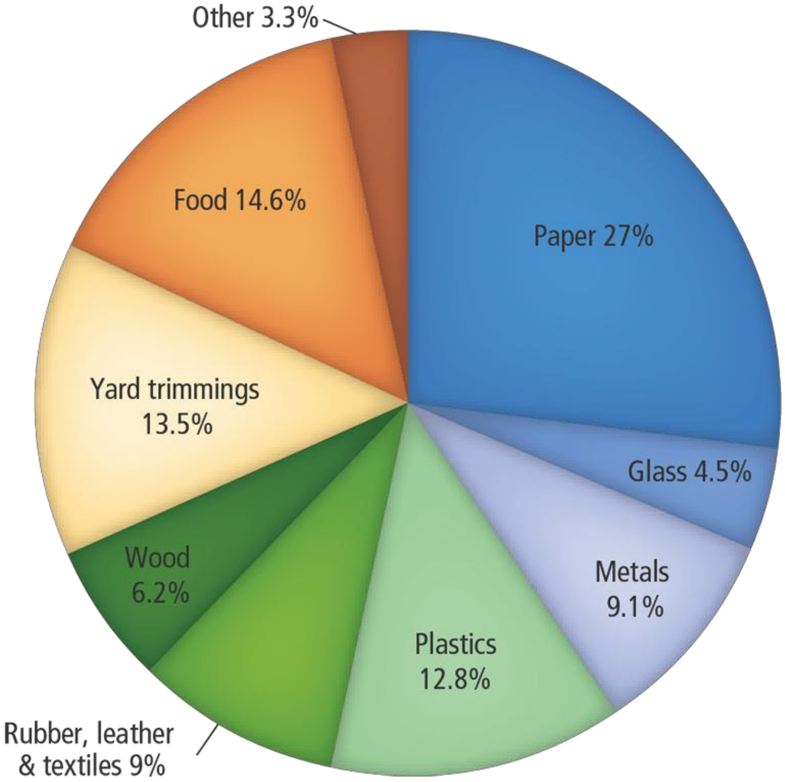


Figure 2.5 MSW composition of US (US EPA, 2013)

2.4 Food Waste Diversion

According to US EPA (2010), “The Food Recovery Hierarchy prioritizes actions organizations can take to prevent and divert wasted food.” Each step of the Food Recovery Hierarchy focuses on different food waste management strategies. The top levels of the hierarchy should be the most preferred ways to prevent and divert wasted food since they are environmentally, socially and economically more beneficial than the rest (USEPA, 2010). Figure 2.6 shows the food recovery hierarchy recommended by US EPA.

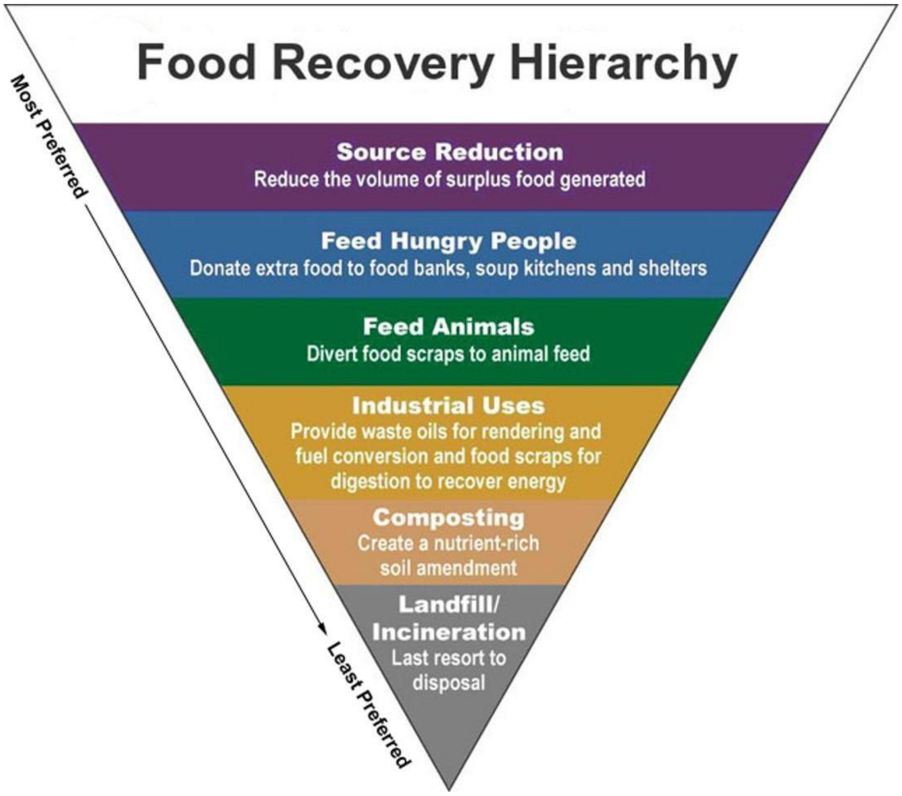


Figure 2.6 Food Waste Hierarchy (US EPA, 2010)

Papargyropoulou et al. (2014) proposed a framework for addressing the food waste challenge. The proposed options and their prioritization were proposed on the basis of environmental and social aspects of food surplus and waste. The three themes considered for the proposed framework were:

a) food surplus, food security and waste

b) avoidable and unavoidable food waste and

c) waste prevention and waste management.

According to Papargyropoulou et al. (2014), the most favorable options occupies the top of the framework and the least favorable options are presented at the bottom of the framework. The prioritization of the options is based on the waste hierarchy. Once the feasible options for prevention are exhausted, the framework proposes avoidable food waste to be primarily recycled into animal feed, or compost as a secondary option. Once recycling becomes improbable, treatment of food waste with energy recovery, such as with anaerobic digestion, is the next preferred option. Disposal in landfill is the least favorable option, when all other options are exhausted (Papargyropoulou et al., 2014). Figure 2.7 outlines the proposed food surplus and waste framework proposed by Papargyropoulou et al. (2014).

Though landfilling and incineration are the least preferred options in the food waste hierarchy, 97% of the food waste goes to incinerator or landfill (US EPA, 2010). An alternative for the waste diversion from landfills is to promote biological treatment of food waste, either by aerobic composting (AC) or anaerobic digestion (AD) (Levis et al., 2010).

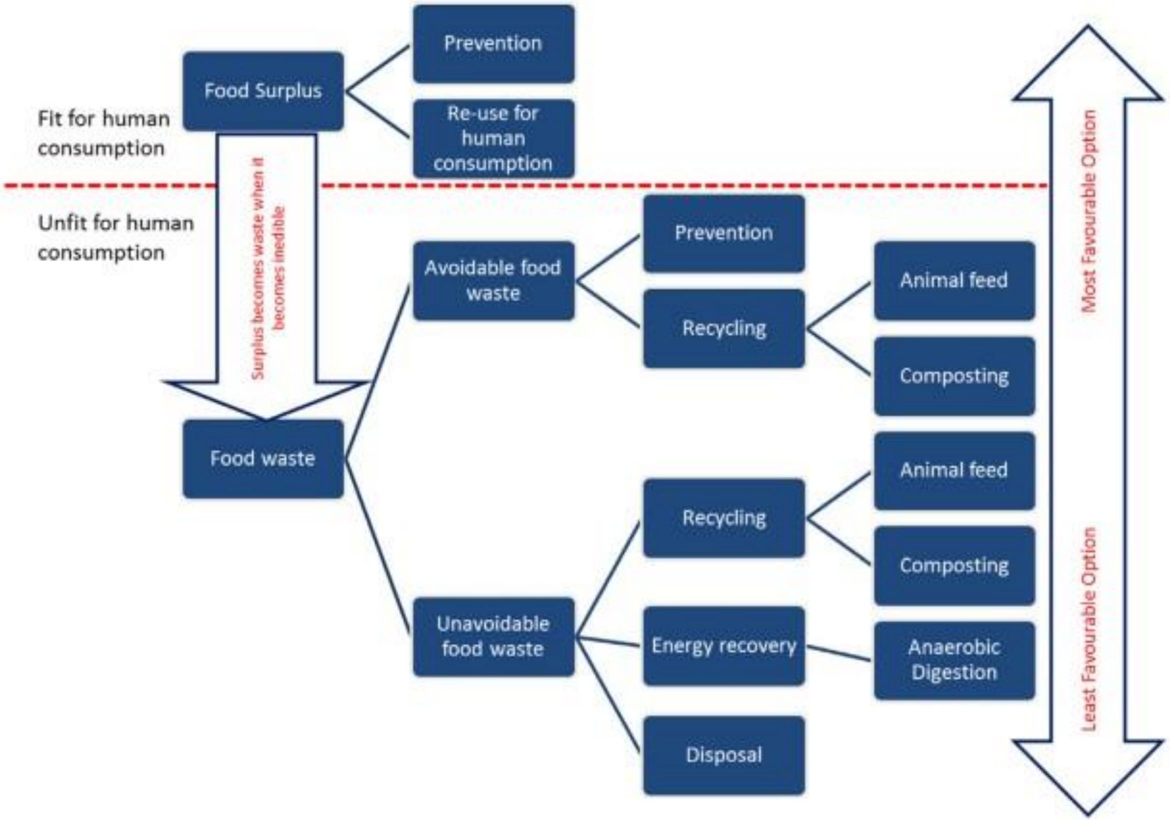


Figure 2.7 Food Surplus and waste framework (Papargyropoulou et al., 2014)

2.5 Organic Waste Ban in Landfills

According to AR News (2014), Connecticut was the first state to ban commercial food waste from landfills in 2011, prohibiting commercial food waste generators (2 or more tons/week) to recycle the materials rather than sending them to a landfill if located within 20 miles of a suitable recycling facility. Vermont also banned commercial food waste from landfills in 2012. By 2020, all types of food waste will be banned from Vermont landfills (AR News, 2014).

Massachusetts is the latest state to ban commercial food from landfills. According to the Massachusetts Department of Environmental Protection (MassDEP, 2011), 90 percent of the 1.4 million tons of organic waste produced is either incinerated or sent to a landfill. Organic waste, after recycling, represents the largest percentage (25%) by volume overall (MassDEP, 2012a). To address this and other issues related to organic waste, MassDEP drafted a Solid Waste Master Plan that seeks to divert food and other organic materials such as compostable paper from the solid waste stream. One stated objective is to divert at least 350,000 tons, or 35 percent, of food waste by 2020 (MassDEP, 2013a). Affected institutions includes large-scale restaurants, hotels, hospitals, colleges and universities, elder care centers, supermarkets, correctional facilities, and food manufacturers. There are approximately 3,000 of these qualifying institutions in Massachusetts (MassDEP, 2012b). This trend is anticipated to continue in future and more and more states may adopt organic waste ban in landfills (USCC, 2015). Figure 2.8 shows the states that ban organics or mandate organic recycling.

🟩 Ban/mandate some yard debris: Arkansas*, Delaware, Florida*, Georgia*, Illinois, Indiana, Iowa, Maryland, Massachusetts, Michigan, Minnesota, Missouri, Nebraska*, New Hampshire, New Jersey, North Carolina, Ohio, Pennsylvania, Rhode Island, South Carolina, South Dakota, Vermont, West Virginia, Wisconsin

*Allow yard debris disposal in landfills that generate energy

🟨 Ban/mandate food scraps: California, Connecticut, Massachusetts, Rhode Island, Vermont.
Also of note: New York City, Seattle

Source: Haaren, Themelis and Goldstein, State of Garbage in America, BioCycle Magazine, Oct 2010, updated 5-2011, 3-2012, 4-2013, 6-2014, 10-2014

Figure 2.8 US States that ban or mandate recycling of yard debris and food waste (USCC, 2015)

2.6 Potential resource recovery from food waste

2.6.1 Types of Food Waste

Food wastes contain complex ingredients, which can be separated from the original material. The varieties of food waste originating from different branches of the food industry can be divided in two main groups (Galanakis, 2012):

i) Plant

ii) Animal

The food wastes can further be divided into 7 subcategories: (i) cereals, (ii) roots and tubers, (iii) oil crops and pulses, (iv) fruits and vegetables, (v) meat products, (vi) fish and seafood and (vii) dairy products (Galanakis, 2012).

2.6.2 High Added Value Component Recovery

According to Galanakis (2012), food waste contains valuable functional compounds derived from agricultural and food processing by-products, which can be recovered. The extraction, fractionation and isolation of these compounds from food wastes require principles of analytical chemistry. The applied methodologies are introduced with following objectives:

(a) maximizing the yield of the target compounds

(b) suiting the demands of industrial processing

(c) clarifying the high added-value ingredients from impurities and toxic compounds

(d) avoiding deterioration and loss of functionality during processing and

(e) ensuring the food grade nature of the final product.

Galanakis (2012) classified the recovery stages in five distinct categories, although some steps are sometimes eliminated or overlap each other. Processing progresses from the macroscopic to the macromolecular level, followed by extraction of specific micro-molecules with subsequent purification and encapsulation of the target compounds (Figure 2.9). This downstream scheme is applied when recovery of two different ingredients or a micromolecule is attempted. However, when the target compound is only a macromolecule (i.e. protein), the separation stage may be omitted (Galanakis, 2012).

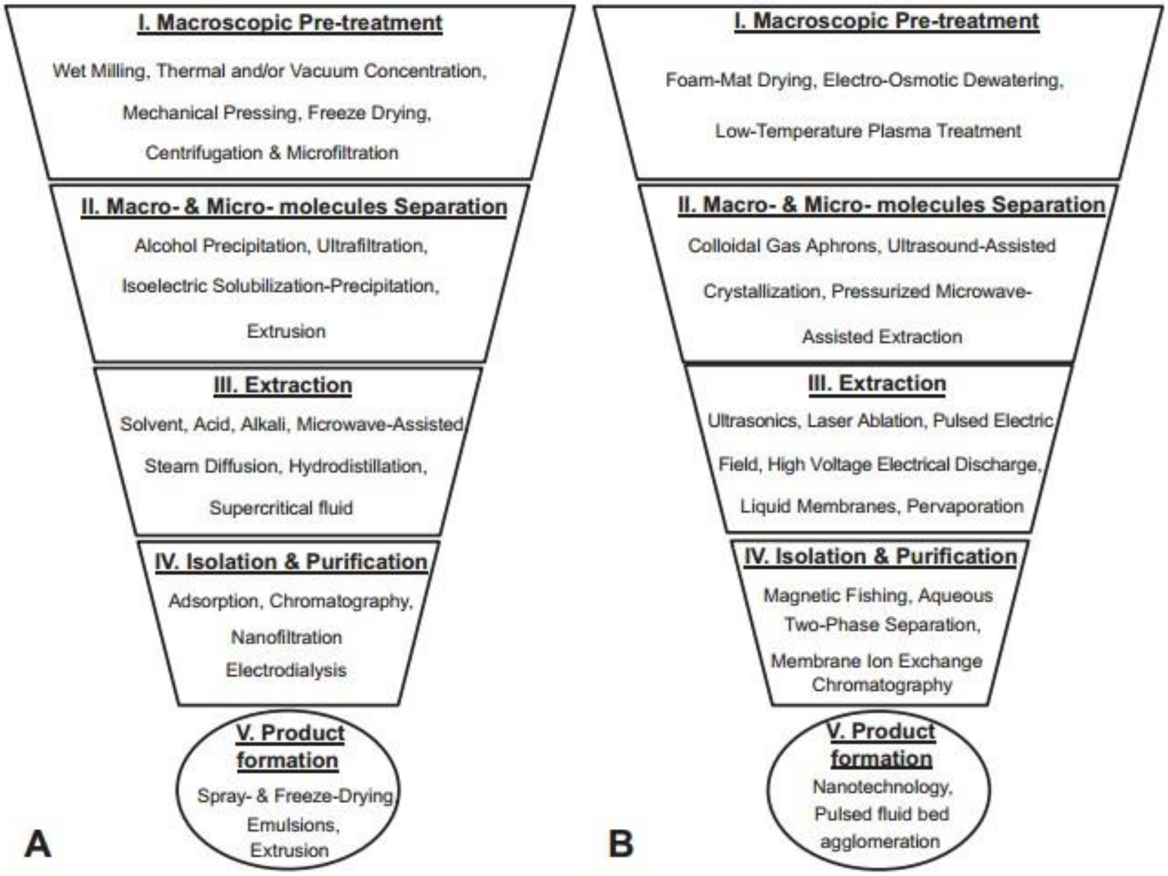


Figure 2.9 Recovery stages of high-added value components from food wastes: (A) Established and (B) emerging technologies (Galanakis, 2012)

Although the recovery and yield of the technologies are important, the product safety and the general cost govern the final decision for methodology selection. These are the critical aspects for emerging technologies, as they could be too sophisticated compared to the expected yield. Conventional technologies (i.e. ultra-filtration and alcohol precipitation) for macro and micro-molecules separation are both safe and cheap. On the other hand, colloidal gas aphrons, despite being a cheap technique, is dependent on the use of biodegradable and non-toxic surfactants for operational safety. Ultrasound- assisted crystallization and pressurized microwave-assisted extraction are considered as green and safe technologies, but the investment cost of the latter is much higher.

Mirabella et al. (2014) analyzed potential value added by-products that can be obtained from different types of food waste. Figure 2.10 A and Figure 2.10 B summarizes all the valuable derivatives from analyzed fruits, vegetables, meat and dairy products.

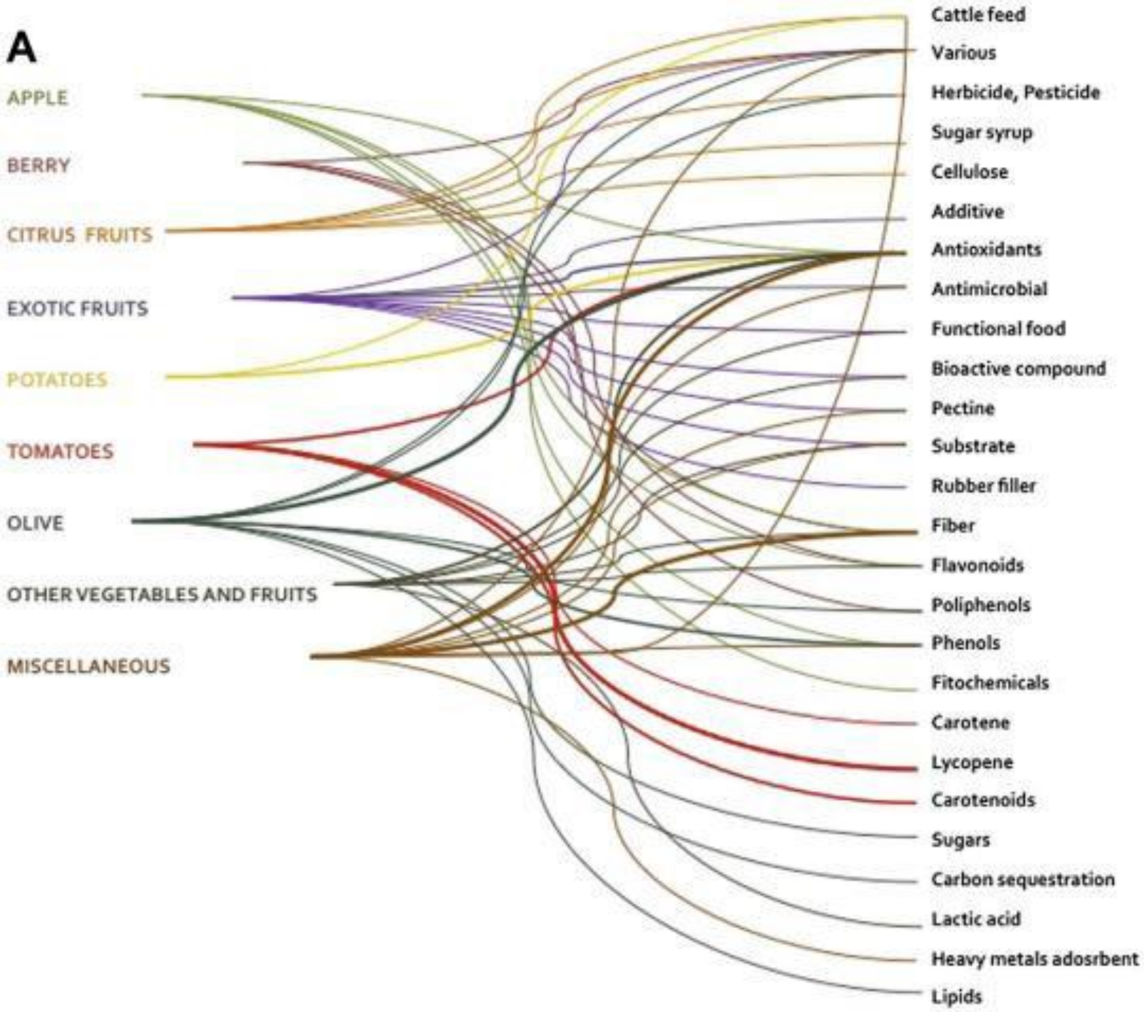


Figure 2.10 A. Summary of valuable compounds derivable from fruits and vegetables analyzed.

(Mirabella et al., 2014) (contd.)

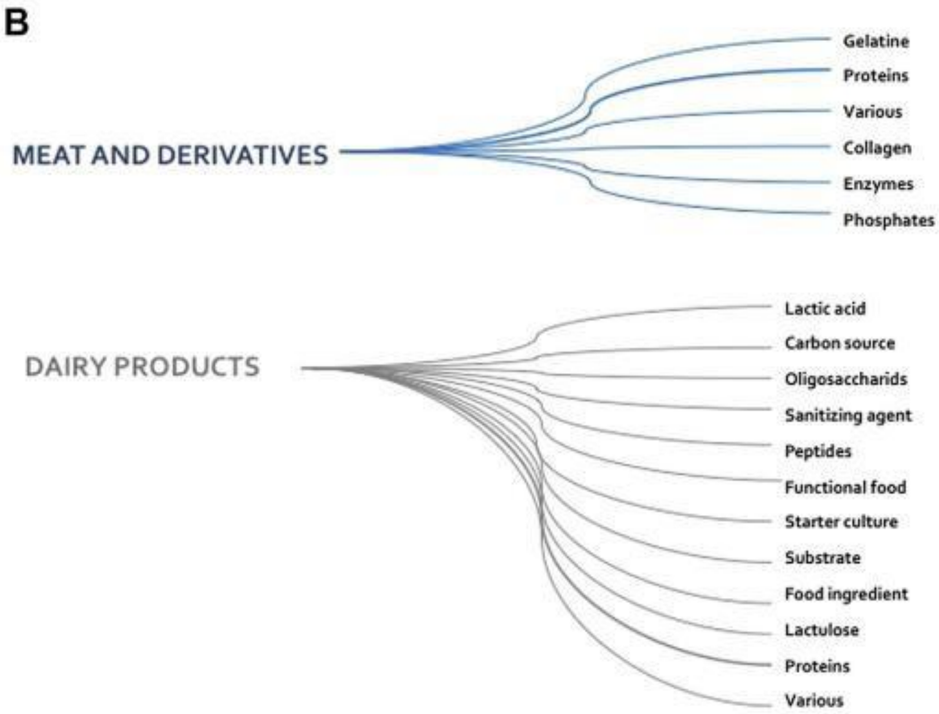


Figure 2.10 B. Summary of valuable compounds derivable from meat and derivatives and dairy products (Mirabella et al., 2014)

According to Mirabella et al. (2014), antioxidants, fiber, phenols, polyphenols and cartenoids are mostly researched for extraction potentials due to high possibility of application. Also, food wastes require intricate processing and incur high research costs before the recovered resources can be used. Therefore, a thorough investigation of type and amount of waste, exploitation potential and potential end users is necessary to justify the investment. Moreover, the environmental impacts of new production processes must be considered for sustainability. Finally, potential health concerns of consumers should be addressed due to excessive modifications of food (Mirabella et al., 2014).

2.6.3 Energy Recovery from Food Waste (FW)

The food waste contained approximately 2030 ± 160 trillion BTU of embedded energy in 2007, which was about 2% of annual energy consumption in US (Cuéllar and Webber, 2010). Food waste is generally incinerated with other wastes for heat or electricity generation (Kiran et al., 2014). However, due to high moisture in FW, dioxins may be produced and incinerating food waste may cause air pollution and loss of chemical value (Katami et al., 2004; Ma et al., 2009).

Potential FW processing includes production methane, hydrogen, ethanol, enzymes, organic acid, biopolymers and bioplastics (Kiran et al., 2014). Utilization of FW for production of organic acids and bioplastics brings about most value ($1000/ton), followed by fuel applications ($200–400/ton), animal feed ($70–200/ton) and electricity generation ($60-150/ton) (Sanders et al., 2007). However, biofuel production is preferred due to low market demand of organic acids and bioplastics (Tuck et al., 2012).

According to Kiran et al. (2014), the following types of biofuels are produced from FW:

i) Ethanol
ii) Hydrogen
iii) Methane and
iv) Biodiesel

2.6.3.1 Ethanol Production

Recently, bioethanol production from cheap feedstock like FW is becoming popular (Lundgren and Hjertberg, 2010). Ethanol, which has a market demand of over 140 million tons per year, serves as a feedstock for polyethylene and other plastic production (Kiran et al., 2014). FW does not require much pre-treatment for ethanol production (Tang et al., 2008). Rather, direct utilization of fresh and wet food waste is preferred over dried food waste due to decreased specific surface area and reaction efficiency of dried substrate and enzyme-substrate reaction respectively (Kiran et al., 2014). Various strategies, including use of high ethanol tolerance strains and cell recycle through sedimentation and membrane retention have been proven to improve ethanol yield significantly (He et al., 2009; He at al., 2012; Wang and Lin, 2010; Wang et al., 2012).

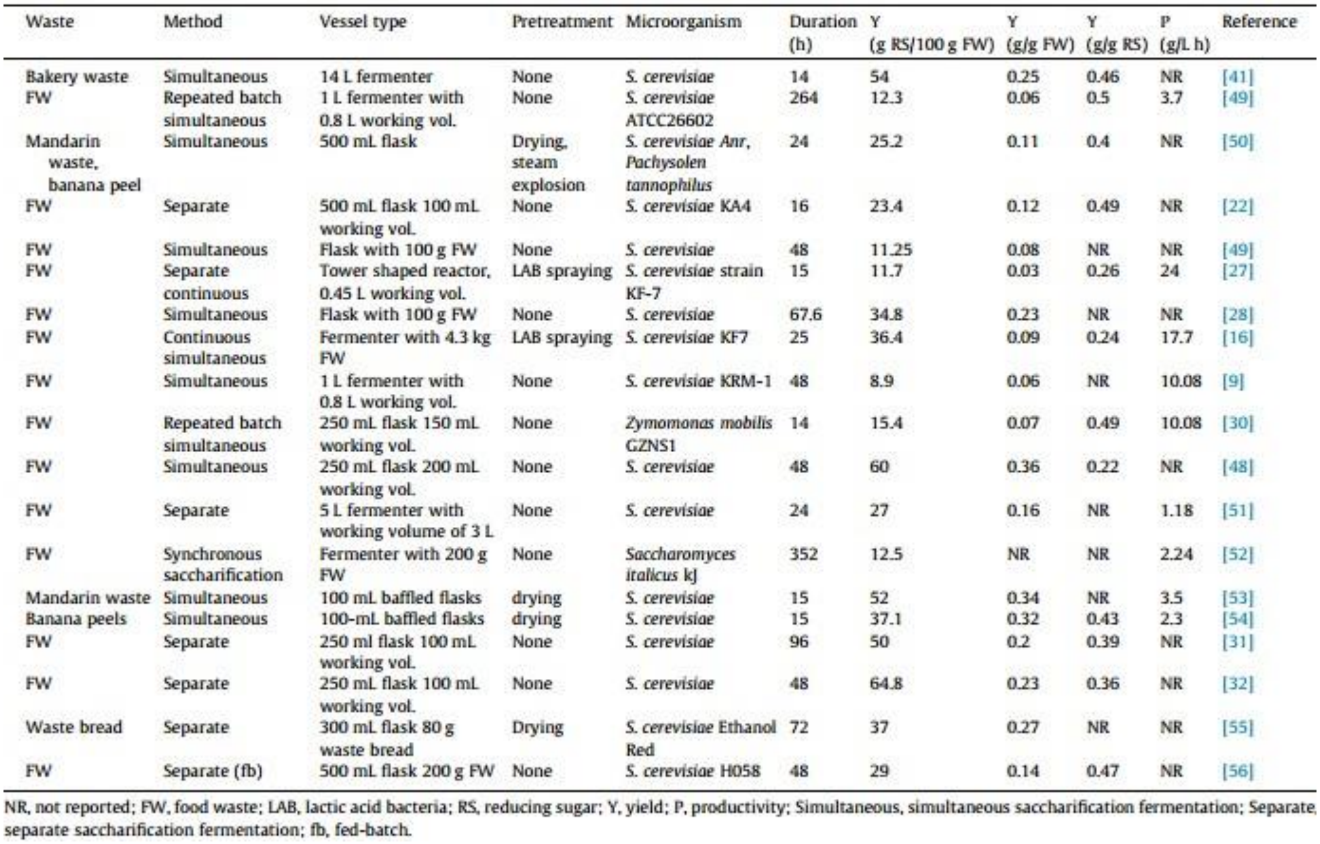
According to Kiran et al. (2014), many pilot and full scale ethanol production plants are operating using various types of wastes as feedstock (Table 2). The estimated annual production of ethanol in South East Asia, Asia and the world are 36.2, 126.8 and 593 thousand billion liters respectively.

2.6.3.2 Hydrogen (H_2) Production

According to Kiran et al. (2014), hydrogen has a high energy yield (142.35 kJ/g) and carbohydrate rich FW is suitable for H_2 production. FW composition, pre-treatment and process configurations may influence H_2 production.

Carbohydrate-based FW has 20 times more hydrogen production potential than fat-based and protein based FW (Show et al., 2012). Pre-treatment of substrate with heat (Heat treatment) suppresses lactate production and increases H_2/butyrate production, with increased cost for large scale operations (Kiran et al., 2014). However, the increased H_2 production due to pre-treatment is short lived and is not very significant (Wang and Zhao, 2009; Luo et al., 2010). Anaerobic sequencing batch reactors (ASBR) and upflow anaerobic sludge blanket (UASB) reactors have greater hydrogen production potential due to their high reactor biomass concentration (Kim et al., 2009). However, only 33% of COD of FW can be harvested as H_2, which is significantly lower than its theoretical value (Kim and Kim, 2013). To achieve higher economic feasibility, combined production of H_2, methane, organic acids and ethanol from FW is recommended (Lin et al., 2013).

Table 2 Ethanol production from food waste (Kiran et al., 2014)



Waste	Method	Vessel type	Pretreatment	Microorganism	Duration (h)	Y (g RS/100 g FW)	Y (g/g FW)	Y (g/g RS)	P (g/L h)	Reference
Bakery waste	Simultaneous	14 L fermenter	None	*S. cerevisiae*	14	54	0.25	0.46	NR	[41]
FW	Repeated batch simultaneous	1 L fermenter with 0.8 L working vol.	None	*S. cerevisiae* ATCC26602	264	12.3	0.06	0.5	3.7	[49]
Mandarin waste, banana peel	Simultaneous	500 mL flask	Drying, steam explosion	*S. cerevisiae Anr, Pachysolen tannophilus*	24	25.2	0.11	0.4	NR	[50]
FW	Separate	500 mL flask 100 mL working vol.	None	*S. cerevisiae* KA4	16	23.4	0.12	0.49	NR	[22]
FW	Simultaneous	Flask with 100 g FW	None	*S. cerevisiae*	48	11.25	0.08	NR	NR	[49]
FW	Separate continuous	Tower shaped reactor, 0.45 L working vol.	LAB spraying	*S. cerevisiae* strain KF-7	15	11.7	0.03	0.26	24	[27]
FW	Simultaneous	Flask with 100 g FW	None	*S. cerevisiae*	67.6	34.8	0.23	NR	NR	[28]
FW	Continuous simultaneous	Fermenter with 4.3 kg FW	LAB spraying	*S. cerevisiae* KF7	25	36.4	0.09	0.24	17.7	[16]
FW	Simultaneous	1 L fermenter with 0.8 L working vol.	None	*S. cerevisiae* KRM-1	48	8.9	0.06	NR	10.08	[9]
FW	Repeated batch simultaneous	250 mL flask 150 mL working vol.	None	*Zymomonas mobilis* GZNS1	14	15.4	0.07	0.49	10.08	[30]
FW	Simultaneous	250 mL flask 200 mL working vol.	None	*S. cerevisiae*	48	60	0.36	0.22	NR	[48]
FW	Separate	5 L fermenter with working volume of 3 L	None	*S. cerevisiae*	24	27	0.16	NR	1.18	[51]
FW	Synchronous saccharification	Fermenter with 200 g FW	None	*Saccharomyces italicus* kJ	352	12.5	NR	NR	2.24	[52]
Mandarin waste	Simultaneous	100 mL baffled flasks	drying	*S. cerevisiae*	15	52	0.34	NR	3.5	[53]
Banana peels	Simultaneous	100-mL baffled flasks	drying	*S. cerevisiae*	15	37.1	0.32	0.43	2.3	[54]
FW	Separate	250 ml flask 100 mL working vol.	None	*S. cerevisiae*	96	50	0.2	0.39	NR	[31]
FW	Separate	250 mL flask 100 mL working vol.	None	*S. cerevisiae*	48	64.8	0.23	0.36	NR	[32]
Waste bread	Separate	300 mL flask 80 g waste bread	Drying	*S. cerevisiae* Ethanol Red	72	37	0.27	NR	NR	[55]
FW	Separate (fb)	500 mL flask 200 g FW	None	*S. cerevisiae* H058	48	29	0.14	0.47	NR	[56]

NR, not reported; FW, food waste; LAB, lactic acid bacteria; RS, reducing sugar; Y, yield; P, productivity; Simultaneous, simultaneous saccharification fermentation; Separate, separate saccharification fermentation; fb, fed-batch.

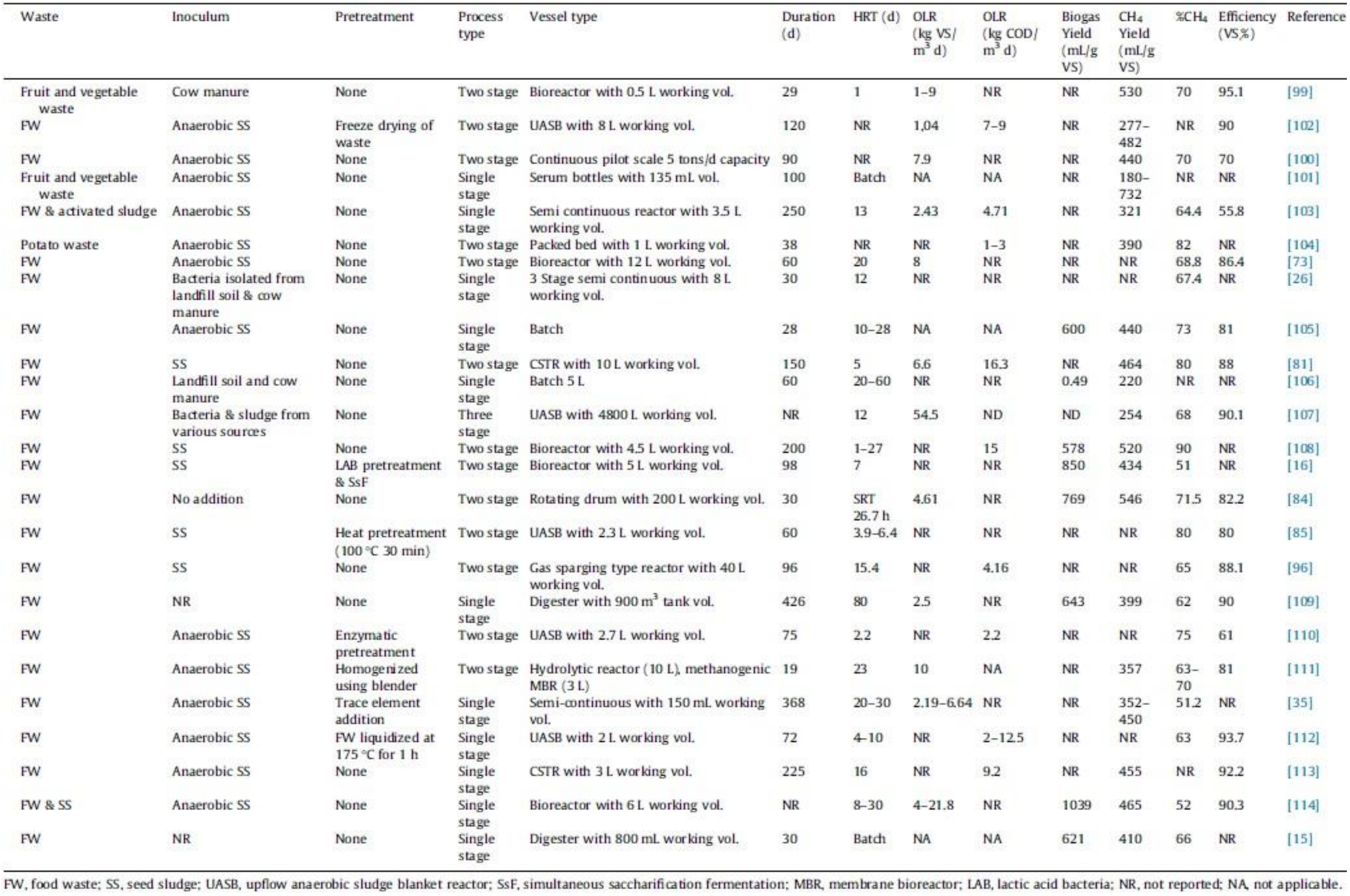
2.6.3.3 Methane Production

Methane production through anaerobic digestion is a feasible option for waste management due to its low cost, utilization as renewable energy and utilization of residual by-product (digestate) as fertilizer or soil conditioner (Kiran et al., 2014; Morita and Sasaki, 2012; Nasir et al., 2012). A wide spectrum of bioconversion (FW to methane) efficiency has been reported, from 70% - 95% Volatile Solids (VS) conversion to 180 – 732 mL/g VS methane yield (Gunaseelan, 2004; Lee et al., 1999; Viturtia et al., 1989). Table 3 lists the studies on methane production by anaerobic digestion from various FWs.

2.6.3.4 Biodiesel production

Food waste can be transformed into fatty acids and biodiesel by direct transesterification using acid or alkaline catalysts or by transesterification of microbial oils obtained from various oleaginous micro-organisms (Chen et al., 2009; Papanikolaou et al., 2011). Also, FW hydrolysate can be used as culture medium and nutrient source for microalgae cultivation, which in turn helps in biodiesel production (Pleissner et al., 2013). With an estimated maximum lipid yield of 0.74 g/g, 647 kilotons of biodiesel can be produced worldwide annually, which can potentially generate 24.5 million GJ energy per year (Kiran et al., 2014).

Table 3 Methane production from food waste (Kiran et al., 2014)



Waste	Inoculum	Pretreatment	Process type	Vessel type	Duration (d)	HRT (d)	OLR (kg VS/m³ d)	OLR (kg COD/m³ d)	Biogas Yield (mL/g VS)	CH_4 Yield (mL/g VS)	%CH_4	Efficiency (VS%)	Reference
Fruit and vegetable waste	Cow manure	None	Two stage	Bioreactor with 0.5 L working vol.	29	1	1–9	NR	NR	530	70	95.1	[99]
FW	Anaerobic SS	Freeze drying of waste	Two stage	UASB with 8 L working vol.	120	NR	1,04	7–9	NR	277–482	NR	90	[102]
FW	Anaerobic SS	None	Two stage	Continuous pilot scale 5 tons/d capacity	90	NR	7.9	NR	NR	440	70	70	[100]
Fruit and vegetable waste	Anaerobic SS	None	Single stage	Serum bottles with 135 mL vol.	100	Batch	NA	NA	NR	180–732	NR	NR	[101]
FW & activated sludge	Anaerobic SS	None	Single stage	Semi continuous reactor with 3.5 L working vol.	250	13	2.43	4.71	NR	321	64.4	55.8	[103]
Potato waste	Anaerobic SS	None	Two stage	Packed bed with 1 L working vol.	38	NR	NR	1–3	NR	390	82	NR	[104]
FW	Anaerobic SS	None	Two stage	Bioreactor with 12 L working vol.	60	20	8	NR	NR	NR	68.8	86.4	[73]
FW	Bacteria isolated from landfill soil & cow manure	None	Single stage	3 Stage semi continuous with 8 L working vol.	30	12	NR	NR	NR	NR	67.4	NR	[26]
FW	Anaerobic SS	None	Single stage	Batch	28	10–28	NA	NA	600	440	73	81	[105]
FW	SS	None	Two stage	CSTR with 10 L working vol.	150	5	6.6	16.3	NR	464	80	88	[81]
FW	Landfill soil and cow manure	None	Single stage	Batch 5 L	60	20–60	NR	NR	0.49	220	NR	NR	[106]
FW	Bacteria & sludge from various sources	None	Three stage	UASB with 4800 L working vol.	NR	12	54.5	ND	ND	254	68	90.1	[107]
FW	SS	None	Two stage	Bioreactor with 4.5 L working vol.	200	1–27	NR	15	578	520	90	NR	[108]
FW	SS	LAB pretreatment & SsF	Two stage	Bioreactor with 5 L working vol.	98	7	NR	NR	850	434	51	NR	[16]
FW	No addition	None	Two stage	Rotating drum with 200 L working vol.	30	SRT 26.7 h	4.61	NR	769	546	71.5	82.2	[84]
FW	SS	Heat pretreatment (100 °C 30 min)	Two stage	UASB with 2.3 L working vol.	60	3.9–6.4	NR	NR	NR	NR	80	80	[85]
FW	SS	None	Two stage	Gas sparging type reactor with 40 L working vol.	96	15.4	NR	4.16	NR	NR	65	88.1	[96]
FW	NR	None	Single stage	Digester with 900 m^3 tank vol.	426	80	2.5	NR	643	399	62	90	[109]
FW	Anaerobic SS	Enzymatic pretreatment	Two stage	UASB with 2.7 L working vol.	75	2.2	NR	2.2	NR	NR	75	61	[110]
FW	Anaerobic SS	Homogenized using blender	Two stage	Hydrolytic reactor (10 L), methanogenic MBR (3 L)	19	23	10	NA	NR	357	63–70	81	[111]
FW	Anaerobic SS	Trace element addition	Single stage	Semi-continuous with 150 mL working vol.	368	20–30	2.19–6.64	NR	NR	352–450	51.2	NR	[35]
FW	Anaerobic SS	FW liquidized at 175 °C for 1 h	Single stage	UASB with 2 L working vol.	72	4–10	NR	2–12.5	NR	NR	63	93.7	[112]
FW	Anaerobic SS	None	Single stage	CSTR with 3 L working vol.	225	16	NR	9.2	NR	455	NR	92.2	[113]
FW & SS	Anaerobic SS	None	Single stage	Bioreactor with 6 L working vol.	NR	8–30	4–21.8	NR	1039	465	52	90.3	[114]
FW	NR	None	Single stage	Digester with 800 mL working vol.	30	Batch	NA	NA	621	410	66	NR	[15]

FW, food waste; SS, seed sludge; UASB, upflow anaerobic sludge blanket reactor; SsF, simultaneous saccharification fermentation; MBR, membrane bioreactor; LAB, lactic acid bacteria; NR, not reported; NA, not applicable.

2.7 Food Waste Treatment in USA: Off-site

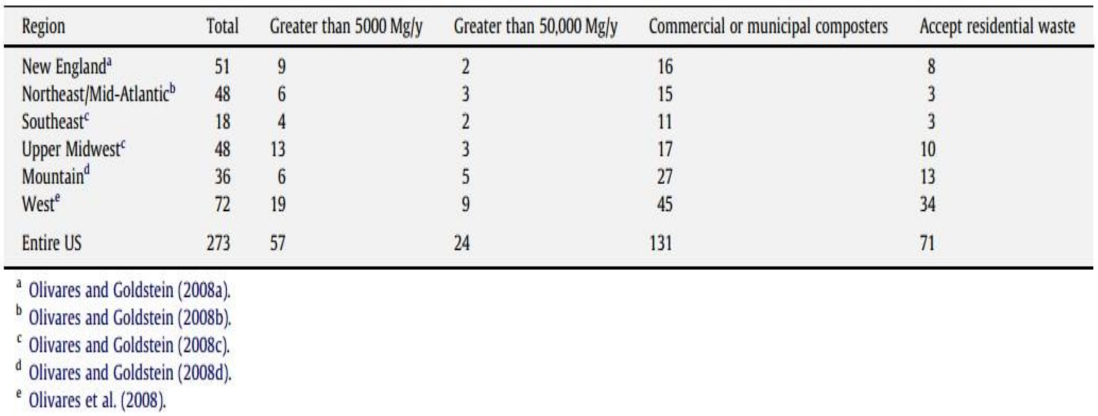
Less than 3% of food waste was recovered and recycled in 2008 and more than 97% ended up in the landfill (US EPA, 2008). According to Levis et al. (2010), an alternative for the waste diversion from landfills is to promote biological treatment of food waste, either by aerobic composting (AC) or anaerobic digestion (AD). While programs and facilities to manage yard waste are well established, food waste management in composting facilities is still a developing practice. However, food waste diversion rate is likely to increase, leading to a considerable interest in food waste composting (Levis et al., 2010).

2.7.1. Aerobic composting (AC) of source separated organics

According to Levis et al. (2010), about 300 facilities in the US accept food waste. Majority (80%) of these facilities process <5000 Mg of food waste per year (100 Mg per week), and less than 10% (< 30 facilities) of these facilities process >50,000 Mg of food waste per year (1000 Mg per week). Most food waste composting facilities in the US are commercial or municipal facilities, followed by colleges, universities and farms. However, only a quarter of the facilities accept residential food waste, with the vast majority (about 75%) processing Industrial-Commercial-Institutional (ICI) wastes. According to Levis et al. (2012), "It is the ICI generators that are currently driving food waste diversion in the US". Table 4 provides a summary of the state of food waste composting facilities in the US.

The compost quality and market availability were a critical parameter for the composting facilities. Current estimates indicate that <3% of US food waste generation is currently treated by composting. There is potential for the quantity of product to increase manifold since less than 3% food waste is currently being processed. However, current composting marketing plans should consider the types of products for end users, product packaging and market location. Bagged products can be sold at significantly higher prices than bulk products, but the market for bagged products is likely to saturate more quickly (Levis et al., 2010).

Table 4 Summary of food waste composting facilities in the US (Levis et al., 2012)



Region	Total	Greater than 5000 Mg/y	Greater than 50,000 Mg/y	Commercial or municipal composters	Accept residential waste
New England[a]	51	9	2	16	8
Northeast/Mid-Atlantic[b]	48	6	3	15	3
Southeast[c]	18	4	2	11	3
Upper Midwest[c]	48	13	3	17	10
Mountain[d]	36	6	5	27	13
West[e]	72	19	9	45	34
Entire US	273	57	24	131	71

[a] Olivares and Goldstein (2008a).
[b] Olivares and Goldstein (2008b).
[c] Olivares and Goldstein (2008c).
[d] Olivares and Goldstein (2008d).
[e] Olivares et al. (2008).

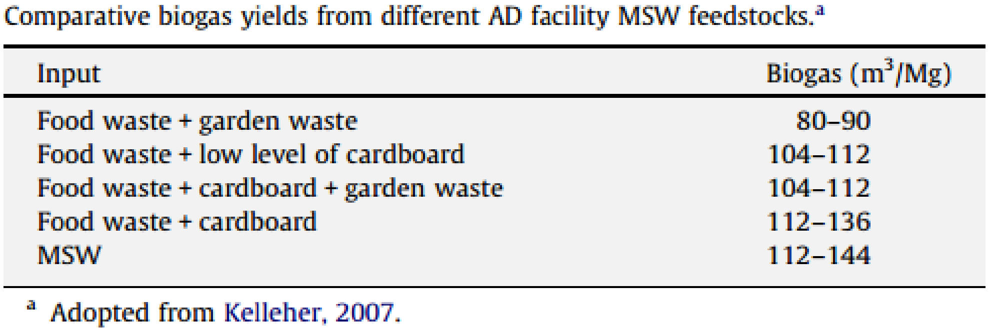
Compost use in horticulture can be greatly enhanced. Composts increase water penetration and retention, improve drought resistance, improve soil tillage properties, build humus content, improve plant health, suppress weeds, and use fewer chemicals (Walker et al., 2006). However, quality concern is the main reason for a decrease in horticultural compost use (Rahmani et al., 2004; Walker et al., 2006). Development and implementation of compost quality standards may increase horticultural compost use. Developing specific blends for local growing and soil conditions can also aid in developing markets for compost. Erosion control and road projects are another major compost use. This is a growing area where government purchases may be able to increase demand (Levis et al., 2010).

2.7.2 Anaerobic digestion (AD) of source separated organics

According to Levis et al. (2010), there is only one medium-scale AD facility operating in North America, with a processing capacity of 42,500 Mg/yr., in contrast to over 120 anaerobic digestion facilities in Europe. The facility, situated in eastern Canada, generates about 110 m^3 of biogas and 0.33 Mg of digested solids per Mg of feedstock with a methane content of 45–73% depending on the intermittent feeding pattern. This amount of methane generates about 10 GWH of electricity annually. Additionally, most of the digestant is sold in bulk at a low price to composting facilities, where the materials are blended with amendments and cured. In summary, as evidenced by the presence of many facilities in Europe, AD is a proven technology for the treatment of food waste (Levis et al., 2010). However, the major limitations of AD implementation are the capital cost and the ability to obtain a pure feedstock. The collection and processing of Source Separated Organics (SSO) has been successfully implemented in many major cities of Canada and the US. However, it is as capital-intensive as a new waste-to-energy (WTE) facility (Levis et al., 2010). Table 5 summarizes the reported biogas yields from different MSW feedstock in anaerobic digestion facilities in Europe (Kelleher, 2007 cited in Levis et al., 2010).

Table 5 Biogas yield from different MSW feedstock in Europe

(Levis et al., 2010)



Comparative biogas yields from different AD facility MSW feedstocks.[a]

Input	Biogas (m^3/Mg)
Food waste + garden waste	80–90
Food waste + low level of cardboard	104–112
Food waste + cardboard + garden waste	104–112
Food waste + cardboard	112–136
MSW	112–144

[a] Adopted from Kelleher, 2007.

2.7.3 Quality and Stability of Compost and Anaerobic digestate from as soil amendment

Biological treatment of organic wastes (AC, AD etc.) serves as efficient methods for waste stabilization and nutrient recovery, with the end product intended for use as soil amendment (Hartmann and Ahring, 2006). Aerobic composting is faster and cost- efficient, but anaerobic digestion has added advantage of energy recovery (Levis and Barlaz, 2011; McDougall et al., 2008). However, the anaerobic digestate is not suitable for soil application without treatment due to phytotoxicity and other associated factors (Abdullahi et al., 2008; McLachlan et al., 2002). Hence, anaerobic digestate obtained from AD processes needs further processing to enhance fertilizer value and applicability as a soil conditioner (Abdullahi et al., 2008). Aerobic treatment of anaerobic digestate enhances fertilizer quality by reducing moisture content, odor and pathogens (Abdullahi et al., 2008; McDougall et al., 2008).

Abdullahi et al. (2008) studied the effects of aerobic and anaerobic post- treatment of organic fractions of MSW on the stability of anaerobic digestate and soil quality using seed germination tests. Seed germination tests indicated that fresh feedstock and digestates at early stages of aerobic post-treatment were phytotoxic. However, phytotoxicity was not observed in soils amended with the fully stabilized anaerobic digestate compost. Also, increase in dilution and incubation time enhanced seed germination and the benefits of Anaerobic Digester Compost (ADC) as soil amendment.

2.7.4 Implications of AC and AD Technologies

According to Levis et al. (2010), numerous factors should be considered in evaluating organic waste management alternatives. Organic wastes produce methane during anaerobic decomposition in landfill or AD facility. The generated methane, if captured, maybe used to generate electricity or steam for heating. These beneficial uses of the methane offset other fuels (e.g., coal and natural gas) cost and impact. However, landfilling, AC and AD processes also use diesel and electrical powered equipment, which cause emissions. These emissions from various processes should be considered in comparing their environmental performances. Also, the soil amendments and composts produced from AC and AD facilities are beneficial as they offset fertilizer production and lead to carbon sequestration. It is more difficult to quantify some of the environmental benefits from land application of AC and AD residuals without Life Cycle Analysis (LCA) (Levis et al., 2010).

Levis and Barlaz (2011) conducted a LCA for commercial food waste processed through aerobic composting systems of varying complexity, anaerobic digestion, and landfills with varying operations. The functional unit of the study was 1000 kg of food waste with 550 kg of branches used as a bulking agent. Global warming potential, NO_x and SO_2 emissions, and total net energy use were determined and compared for each alternative. AD was found to be the most environmentally beneficial treatment option, leading to 395 kg net CO_2 emission per functional unit. AD is favorable mainly because of avoided electricity generation and soil carbon storage from use of the resulting soil amendment. For composting, the use of compost to offset peat has greater emission offsets compared to compost as a fertilizer (Levis and Barlaz, 2011). Table 6 and Figure 2.11 compares the energy wins and environmental impacts of the available food waste treatment alternatives with landfill operations respectively.

Table 6 Comparison of options for biological treatment and optimal capacity ranges

(ISWA 2013)

Food waste options	Annual amount [t] (optimal range)		End-product		Typical energy win
General concept	**Minimum**	**Maximum**	**Type**	**Quality**	**kWh per ton**
Open windrow composting	n.a.	15,000	compost	ready for market	none
In vessel composting	20,000	>200,000	compost	ready for market	none
Wet AD systems	30,000	>200,000	digestate liquid	compostable fertiliser	645
Dry AD systems	20,000	>200,000	digestate	Compostable fertiliser	600
Integrated AD-A	20,000	>200,000	compost	ready for market	510

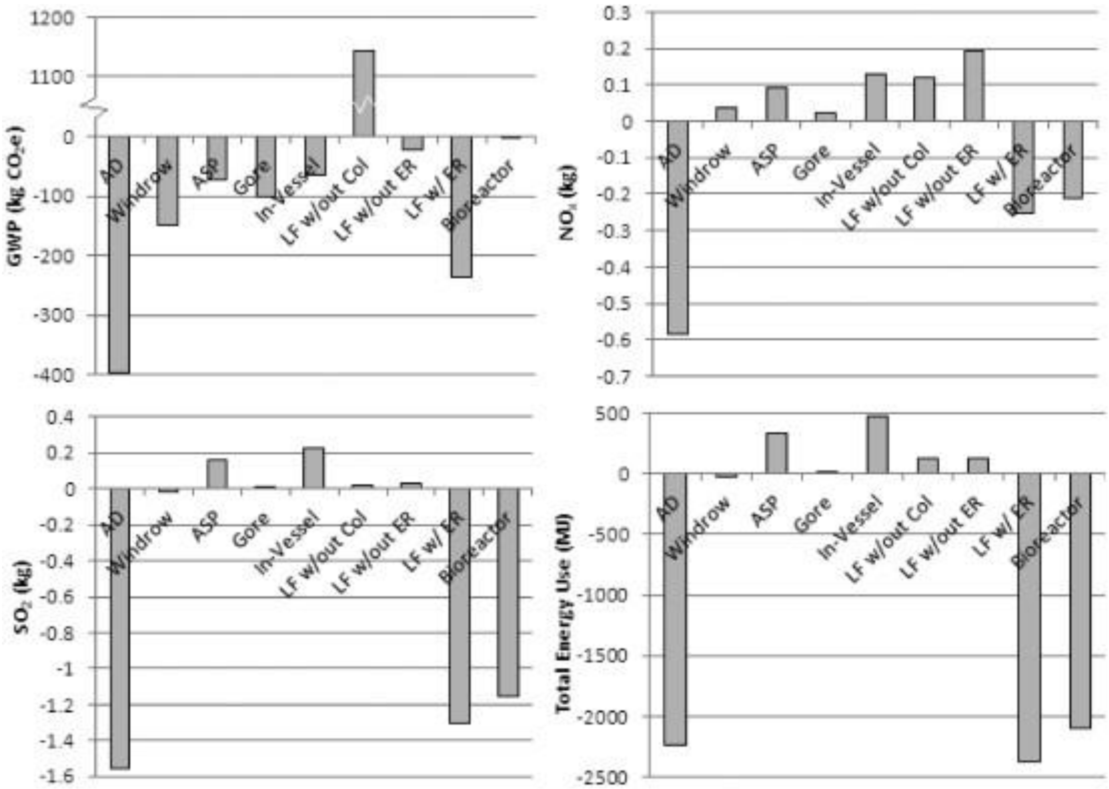


Figure 2.11 GWP, SO_2, NO_x and total energy use for food waste treatment alternatives

(Levis and Barlaz, 2011)

2.8 Food Waste Treatment in USA: On-site

Institutions affected by organic waste ban may choose to have their organic waste hauled away and processed off-site, while others may prefer on-site solutions to manage their food waste (MassDEP, 2013b). According to Cook (2015), there are many emerging technologies presenting themselves as viable alternatives to landfilling food waste. These technologies are both economically and environmentally viable and encompass the limits of food waste management opportunities, from small commercial establishments to municipal sized operations.

Currently, three types of technologies, mentioned below, exist to provide affected institutions with on-site food waste processing options (Cook, 2015; MassDEP, 2013b):

1) Non-biological systems
2) Biological Digesters

2.8.1 Non-Biological systems and End Products

Non-biological systems use mechanical processes to reduce weight and volume of food waste, essentially by removing water. Pulpers/shredders and food waste dehydrators are the two types of non-biological systems commonly available.

Pulpers/shredders use mechanical blades to grind food waste and remove water content. They are capable of processing all sorts of food waste. About 80-90% volume/weight reduction is achieved, based on feedstock composition. The resulting end product consists of a pulp (about 20% of total output) and grey water (about 80% of end product) (Cook, 2015). The semi-wet pulp can be used off-site or as a feedstock for other on-site systems and the grey water can be disposed or used for irrigation (Cook, 2015; MassDEP, 2013b).

Dehydrators combine heat and mechanical agitation processes to evaporate moisture and grind food waste (Cook, 2015). The input materials may contain soiled paper, waxed cardboard and napkins in addition to all categories of food waste (Neale, 2014). The obtained end products comprise of a dry, sterile, odorless pulp (about 80% of total output) and evaporated water (about 20% of total output) (Cook, 2015). The pulp could be used as a composting feedstock, ingredient in animal feed or soil fertilizers with appropriate regulatory approval (Cal Recycle, 2014; MassDEP, 2013b). The evaporated water is collected and can be disposed in sewer or used for irrigation (Cook, 2015; MassDEP, 2013b).

2.8.2 Biological Digesters and End products

Biological digesters can broadly be classified into two types: Anaerobic digesters and Aerobic digesters.

2.8.2.1 Wet/Dry Anaerobic Digester and End Products

Wet/dry anaerobic digesters employ microbes to achieve controlled decomposition of organic materials in absence of oxygen. Anaerobic digesters can accept all organic materials, including paper and compostable bagging, except woody organics (timber, tree branches). The moisture content of the feedstock influences the decision of the system adopted (wet or dry) (Cook, 2015). The end products derived from the process are fertilizer solids (digestate), water and biogas (approximately 70% methane, 28% CO_2) (Cook, 2015). The produced methane powers the whole operation, with the surplus energy available for sale. The remnant digestate is processed for fertilizer or composting, and the waste-water is discharged with necessary treatments (Cook, 2015).

2.8.2.2 Wet aerobic systems and End Products

Wet aerobic systems break down the food waste aerobically (in the presence of oxygen) using natural organic bacteria (Cook, 2015). They accelerate decomposition process through a mixture of ground food waste, water and nutrient mixes (MassDEP, 2013). However, they do not accept all kinds of food waste. Some items like compostable bags, large bones, mussel and clamshells, pineapple tops, cornhusks and raw bread dough cannot be processed with these systems (Cook, 2015). The resulting end product is nutrient-rich wastewater, which is discharged into the municipal wastewater system. Though the manufacturers claim the processed effluent to be safe for discharge, independent tests have indicated levels of biochemical oxygen demand (BOD) above municipal wastewater standards (MassDEP, 2013b).

2.8.2.3 Dry aerobic systems and End Products

Dry aerobic systems, also known as in-vessel composters, break down the input food waste aerobically. Like wet aerobic systems, they are unable to process large bones, mussel and clamshells, pineapple tops, cornhusks and raw bread dough. However, they can process compostable bags (Cook, 2015). Some of the systems may require additional micro-organisms or nutrients to function. The processing time for these systems vary between 24 hours to 14 days. The resulting end product, unlike wet composters, is claimed to be compost. However, the independent tests have found the end-product to be biologically unstable and testing of the produced compost is recommended prior to soil applications (Cook, 2015; MassDEP, 2013b).

While the on-site food waste treatment systems are often successful ways to manage food waste on-site, they are still associated with areas of uncertainty, particularly with regard to the end products of some systems (MassDEP, 2013b). Due to relatively small usage of dehydrators and biodegesters in the US, wastewater authorities and regulatory officials have very limited knowledge of the systems and their output (BioCycle, 2013). The main sustainable benefit of dehydrators is the retention of nutrient values. However, the condensed water from the dehydrator does contain BODs, though not in exceeding amounts. On the other hand, bio digester effluent carry high levels of BOD and sludge and is alarming for wastewater treatment plants (BioCycle, 2013).

Presently, on-site processing of organic waste seem appealing on economic, logistical and hygienic levels. However, detailed knowledge of the processing and end-products are still not available. According to BioCycle (2013), ”Indeed, with every system reviewed and researched there are tangible benefits although none are without costs and opaque product claims. Ultimately, what is clear is that as a result of the light regulatory environment governing dehydration and bio-digestion systems coupled with their relatively new appearance on the market, the customer bears the responsibility to verify and validate vendor claims.”

2.9 Economic Feasibility of Resource Recovery

Adopting sustainable food management practices can help reducing environmental footprint, lowering disposal costs and producing value-added by-products like compost and biogas (Kim, 2014; USEPA, 2016). Aerobic composting produces nutrient-rich compost, which is generally sold at higher prices than commercial fertilizer. AD of food waste produces methane rich biogas, used to produce heat and electricity and digestate, which can be used as a soil amendment (Kim, 2014).

2.9.1 Food waste diversion potential

A study by Goldman and Ogishi (2001) concluded that solid waste disposal and diversion is a major segment of the US economy. Also, as shown in Figure 2.12, the economic impacts of waste diversion are much higher than waste disposal. Waste diversion has been found to be an acceptable alternative both from environmental and economic standpoint (Beck, 2001; Goldman and Ogishi, 2001).

.

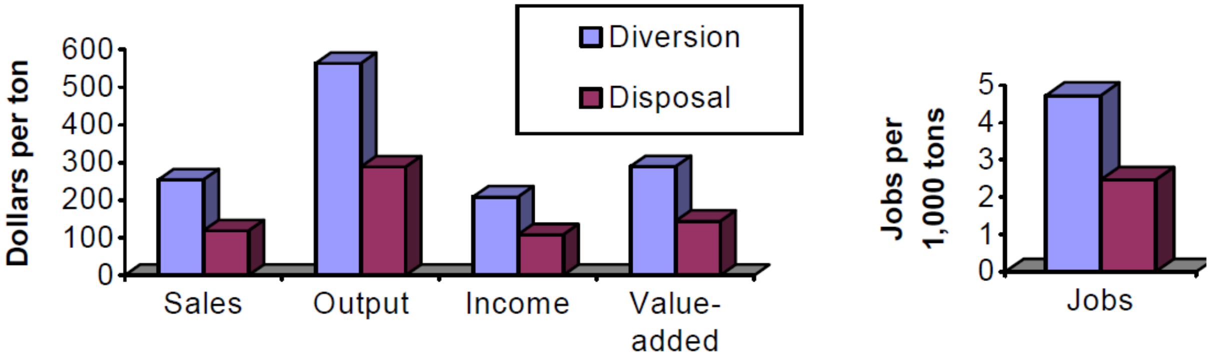


Figure 2.12 Economic impacts of disposal and diversion (per ton) (Goldman and Ogishi, 2001)

According to ReFED (2016), about 13 million tons can be diverted away from the US landfills annually. Figure 2.13 shows the three major diversion alternatives and their diversion and economic potential. Together, these alternatives - Centralized composting, Centralized AD, Waste Water Recovery Facility (WRRF) with AD - can divert 9.5 million tons of food waste annually. However, co-ordination of policy, collection infrastructure and centralized processing facilities are required to realize the potential of these alternatives (ReFED, 2016). Table 7 lists the details of the widely adopted food waste diversion alternatives that can be adopted in the US.

Figure 2.13 Diversion potential and economic value of major food waste diversion alternatives (ReFED, 2016)

Table 7 Potential and economics of Food waste diversion alternatives (Adopted from ReFED, 2016)

Diversion strategy	Diversion Potential (Million	Economic Value (Million $)*	Time Frame	Beneficiary
Centralized Composting	5	18	Medium Term	Municipalities, Compost operators
Centralized AD	1.9	40	Medium Term	Municipalities, AD operators
WRRF with AD	1.6	38	Medium Term	Municipalities, WRRFs
Commercial Greywater	0.595	19	Near Term	Restaurants, foodservice, equipment vendors
Community Composting	0.167	- 6	Near Term	Consumer-facing businesses, consumers, municipalities

Home Composting	0.097	14	Near Term	Consumers
Animal Feed	0.049	- 3	Near Term	Farmers, manufacturers, consumer facing businesses
In-Vessel Composting	0.012	- 1	Near Term	Consumer-facing businesses

*Positive values indicate net benefit whereas negative value indicate net cost

2.9.2 Economics of End products of diversion

About 75% of food waste reduction is possible in US through appropriate recycling or diversion. Centralized Composting and Centralized AD together can potentially support 73% of recycling opportunity, with another 17% contributed by WWRF with AD. The rest of the contribution (10%) is expected from decentralized solutions in homes and businesses. Figure 2.14 shows the framework of recycling solutions and their end products.

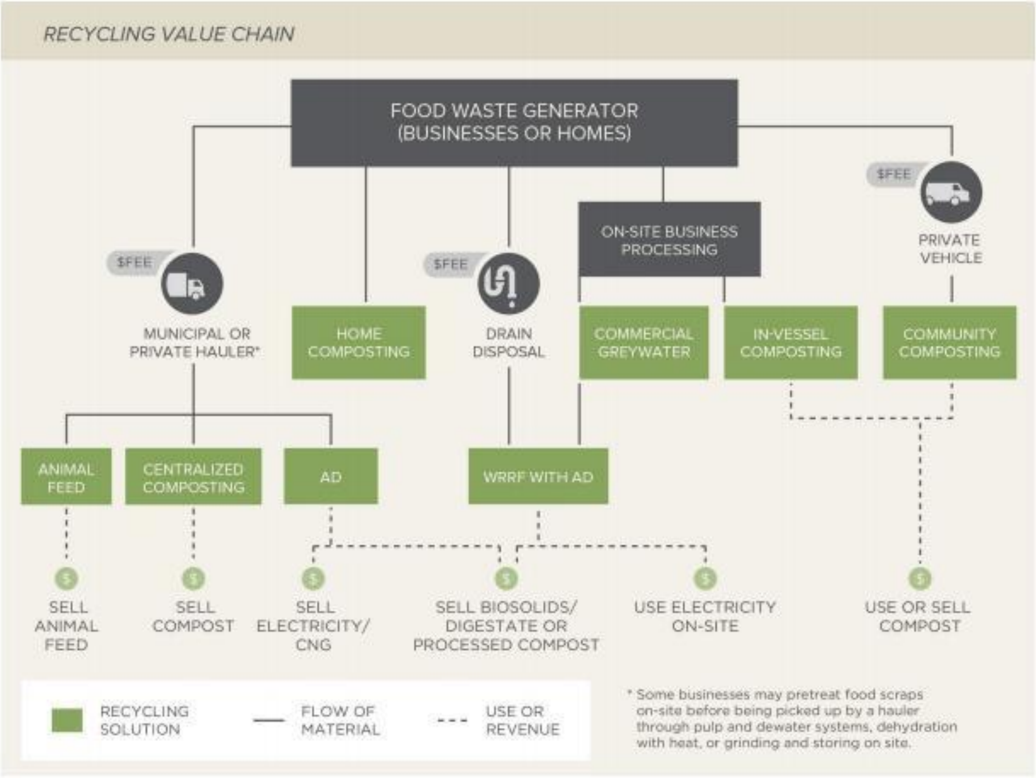


Figure 2.14 Recycling alternatives of food waste and their by-products

Kim (2014) conducted a Cost-Benefit Analysis (CBA) of different scenarios of composting and AD for food waste diversion. Figure 2.15 shows the Net Present Value (NPV) of all the scenarios considered in the study for a 20-year system lifetime. Composting registered the highest due to its lower capital cost. On the other hand, scenario C10 proved most profitable up to a discount rate of 10%. This is due to higher inflation rate of CNG prices. Also, digestate revenue is a critical factor to economic feasibility of AD, since none of the AD scenarios without digestate exhibit positive NPVs.

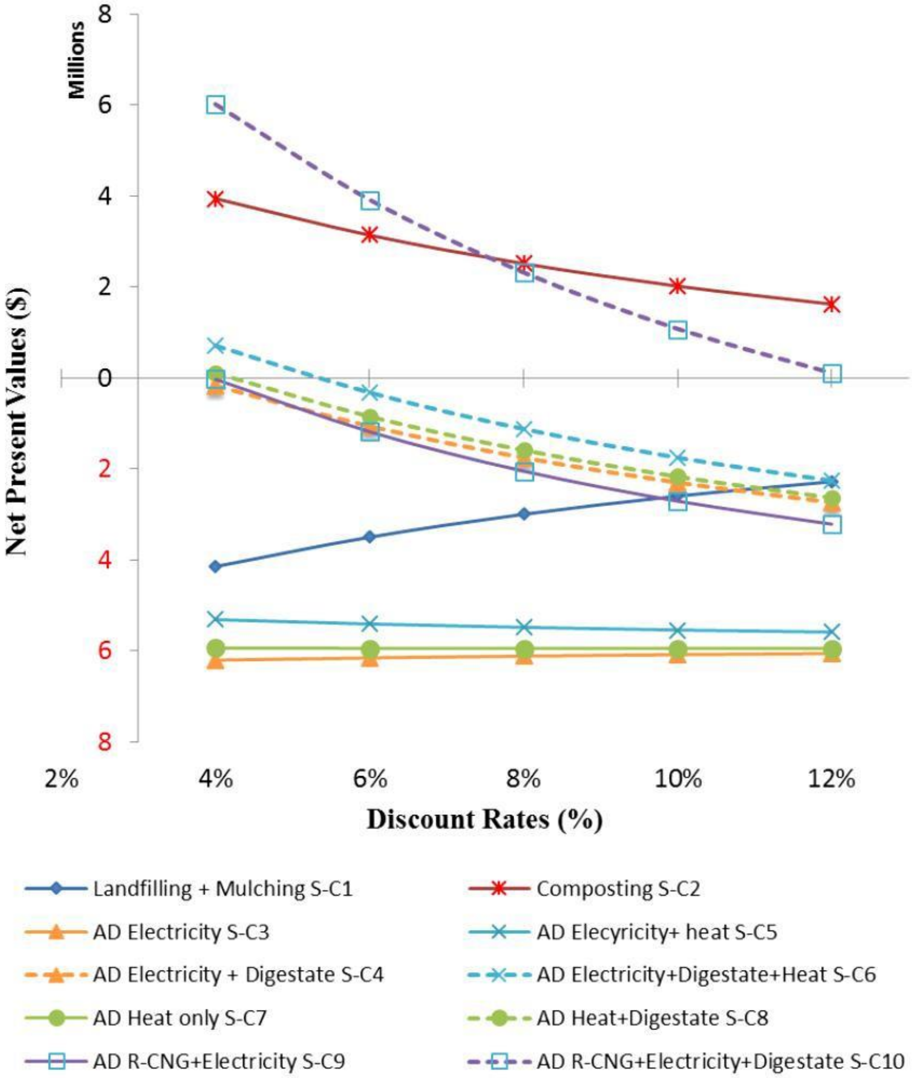


Figure 2.15 NPVs of various diversion scenarios over 20 year lifetime

2.10 Summary

Food waste is the second largest constituent of MSW in USA (US EPA, 2013) and about 1.2 billion metric tons of food is wasted annually across the globe (FAO 2011). More than 97% of the generated food waste is landfilled (US EPA, 2008), though it is the least preferred option in the food waste hierarchy (US EPA, 2010). However, a growing number of US states are banning commercial food wastes in their landfills (AR News, 2014; MassDEP, 2011). Though biological treatment of food waste through aerobic composting (AC) and anaerobic digestion (AD) are well established practices, current processing capacities of these facilities in US is inadequate (Levis et al., 2010). Hence, the ICI generators of food waste are being inclined to prefer on-site solutions to manage their food waste (MassDEP, 2013b).

Among the available on-site treatment systems, food waste dehydrators are non-biological, effective on-site treatment systems which achieve 70-90% volume and mass reduction (Spencer, 2008). The dehydrator output consists of sterile dehydrated organic biomass and condensate water. The suggested uses of the end products are use of the dehydrated organic biomass as soil amendments and condensate water for landscaping or other recoverable use. However, no systematic study was undertaken to evaluate the potential of food waste dehydrators to divert food waste from the landfill and assess the quality of its end products. In the current study, the diversion potential and the quality of dehydrator outputs are assessed, along with the economics of its operation.

Chapter 3

Methodology

3.1 Introduction

The main objective of the present study is to propose a sustainable management strategy of food waste using food waste dehydrators. Accomplishment of the proposed objective required determination of food waste diversion potential from landfill and sustainable usage of the subsequent end products obtained from the food waste dehydrator. To achieve these specific objectives, a set of appropriate laboratory tests were performed on obtained food waste and end products from the food waste dehydrator. This chapter describes the physical, chemical and biological tests performed on food wastes and food waste dehydrator end products, dehydrated food waste (DFW) and condensate water.

Food waste samples were collected from different sources and characterized based on physical composition, moisture content and unit weight of the obtained samples. Food waste dehydrator was operated with the samples to determine the weight loss achieved by the dehydrator. Finally, the obtained end products from the dehydrator (DFW and condensate water) were tested for unit weight, pH, BOD, fecal coliform and enterococci to propose their sustainable usage.

3.2 Food waste collection

Food waste was collected from four different sources – Institutional, household, restaurant and grocery. The collected samples were preserved at 4º C in an environmental growth chamber to prevent the loss of moisture and alteration of other properties of the sample (figure 3.1). Food waste collection from different sources is described as follows:

Figure 3.1 Sample storage in Environmental Growth Chamber at 4º C

3.2.1 Institutional food waste collection

Food scrapings from lunch buffet were collected in a large trash bag from UTA dining (Connection Café). Two samples were collected at different days of the week to get a representative sample from the source and observe the variations. The weight of the samples ranged from 40-60 lbs. Figure 3.2 shows the food waste sample collected from Connection Café.

Figure 3.2 Food Waste Sample collected from Institutional Source (Connection Café)

3.2.2 Household food waste collection

Trash bags were set up at participating houses in the student community and source separated food waste was collected over the week. A total of 2 samples were collected over two consecutive weeks from 7 different participating households to gather a representative sample from the community. Figure 3.3 shows the collected sample from a single household.

Figure 3.3 Food waste collected from a single household

3.2.3 Restaurant food waste collection

Restaurant food waste from the City of Denton is picked up by the collection trucks every morning. To gather a representative sample of restaurant food waste, samples were collected directly from the working face of the landfill. A total of 2 samples were collected every Wednesday for two consecutive weeks. Figure 3.4 shows sample collection and collected samples from the working face of City of Denton landfill.

(a) (b)

Figure 3.4 (a) Collection of restaurant waste from the active face of landfill (b) collected restaurant sample

3.2.4 Grocery food waste collection

Food waste generated in the grocery is delivered to the City of Denton landfill every Wednesday morning. Since grocery food waste samples contain only fruits and vegetables, they are delivered directly to the composting facility rather than working face of the landfill. Two samples were collected from the composting facility every Wednesday for two consecutive weeks. Figure 3.5 shows sample collection and the disposed grocery waste.


(a) (b)

Figure 3.5 (a) Grocery waste at the composting facility in City of Denton landfill, (b) Grocery waste sample collection

3.3 Experimental Program

To meet the objective of proposing a sustainable management strategy of food waste using food waste dehydrators, an extensive experimental program was undertaken. Initially, the obtained food waste samples from four different sources were characterized by their physical properties (physical composition, moisture content and unit weight) before operating the food waste dehydrator with the samples. The end products obtained after dehydrator operation, dehydrated food waste (DFW) and condensate water, are tested for their physical, chemical and biological properties to determine their sustainable usage. The experimental program undertaken in this study is discussed under four subsections: Physical properties of food waste, GAIA food waste dehydrator operation, properties of dehydrated food waste and properties of condensate water. The experimental program is represented by the flow chart shown in Figure 3.6.

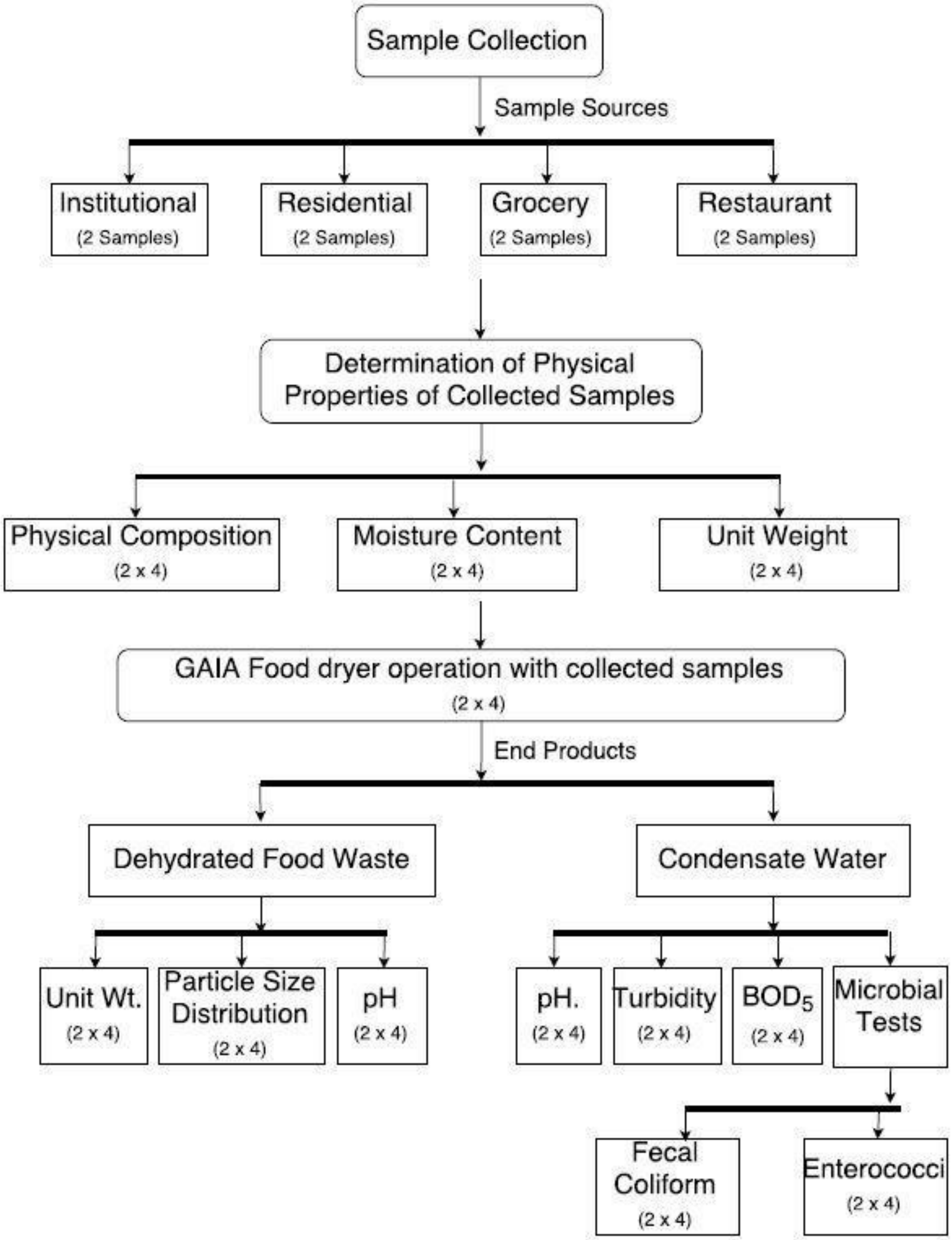


Figure 3.6 Flow chart of Experimental Program

3.4 Physical properties of food waste

3.4.1 Physical composition

The collected food waste samples were manually sorted in five categories: Grains, Fruits and Vegetables, Meat, Seafood and Dairy products. The sorted individual components were weighed to determine their physical composition. Figure 3.7 and Figure 3.8 show sorting of a household food waste sample and the sorted fractions respectively.

Figure 3.7 Manual sorting of food waste samples

(a) (b) (c)

(d) (e)

Figure 3.8 Sorted fractions of food waste sample (a) fruits and vegetables, (b) Grains and cereals, (c) Meat, (d) Dairy and (e) Seafood

3.4.2 Moisture Content

About 2 lbs. of sample was separated from the collected sample before sorting to determine the moisture content of the sample. The separated sample was put in a drying oven, operated at 65º C for 5 days to prevent volatilization of organic content of the food waste. The moisture content of the food waste samples were determined on wet weight basis. Figure 3.9 shows a separated sample of institutional food waste before and after drying.

(a) (b)

Figure 3.9 Institutional Food Waste Sample (a) before drying, (b) after drying

3.4.3 Unit weight:

The unit weight of the obtained food waste samples were measured according to the Standard Proctor Compaction method (ASTM D698) as shown in Figure 3.10. However, a larger proctor mold of volume 1/13.33 cubic feet with detachable collar was used to accommodate different sized waste particles. A hammer weighing 5.5 lbs was dropped 50 times from 12 inch height to impart standard compaction energy per unit volume for one layer. The mold was filled to the rim in three successive layers.

Figure 3.10 Determination of unit weight of food waste

3.5 Food waste dehydrator operation

The GAIA food dryer (Model G-200 H) was used to process the food waste samples. The operation time of the machine ranged from 3 - 4.5 hours, based on the weight of samples processed. Food waste is loaded into the dehydrator from the top loading window and the machine was operated from the control panel provided (Figure 3.11).

Figure 3.11 GAIA food waste dryer loading and operation

After the completion of the dehydration process, the machine is automatically switched off and is ready to discharge the dehydrated food waste. The dehydrated food waste is collected from the output window by operating the control panel, whereas the condensate water is continuously discharged from an outlet at the back of the dryer and is collected in a water container (Figure 3.12).

(a) (b)

Figure 3.12 Collection of dehydrator outputs (a) dehydrated food waste

(b) condensate water

3.6 Properties of dehydrated food waste

The following properties of dehydrated food waste were evaluated to determine its potential usage as a compost feedstock: unit weight, particle size distribution, pH and C:N ratio.

3.6.1 Unit Weight

The unit weight of the dehydrated food waste sample was measured according to the Standard Proctor Compaction method (ASTM D698). Unit weight of the dehydrated food waste samples were determined under both uncompacted and compacted (standard compaction) conditions. For uncompacted condition, a mold of volume 1/13.33 cubic feet, with 6 inch diameter and about 4.6 inch height, was filled without any compaction effort. For standard compaction, the same mold was used (Figure 3.13). A hammer weighing 5.5 lbs was dropped 50 times from 12 inch height to impart standard compaction energy per unit volume for one layer. The mold was filled to the rim in three successive layers.

Figure 3.13 Determination of unit weight of dehydrated food waste

3.6.2 Particle size distribution

The dehydrated food waste obtained from GAIA Food Waste Dryer was sieved to determine their particle size distribution. Sieves ranging from 1 inch opening to No.200 sieve were arranged in descending order and weighed. Then, the samples were put on the top of the sieve set and sieved for 10 minutes. The fractions retained on each sieve was weighed to determine the particle size distribution of the sample. Figure 3.14 shows the sieve setup and samples retained on the sieves.

(a)

(b)

(c)

(d)

Figure 3.14 (a) Sieve setup on mechanical shaker, (b) fraction retained on 1’’ sieve, (c) fraction retained on No. 10 sieve and (d) fraction retained on No. 60 sieve

3.6.3 pH

pH of dehydrated food waste was determined by reading the pH of a 1:5 (DFW: Distilled water) solution, according to US Composting Council (USCC) recommendation. For this test, bench top Oakton pH meter was used. The pH meter is calibrated using buffer solutions of known pH to ensure precise reading. 5 gm of oven dried samples was mixed with 25 ml of distilled water and mixed thoroughly for 10 seconds. The mixture was allowed to stand for 10 minutes and pH of the mixture was recorded (Figure 3.15).

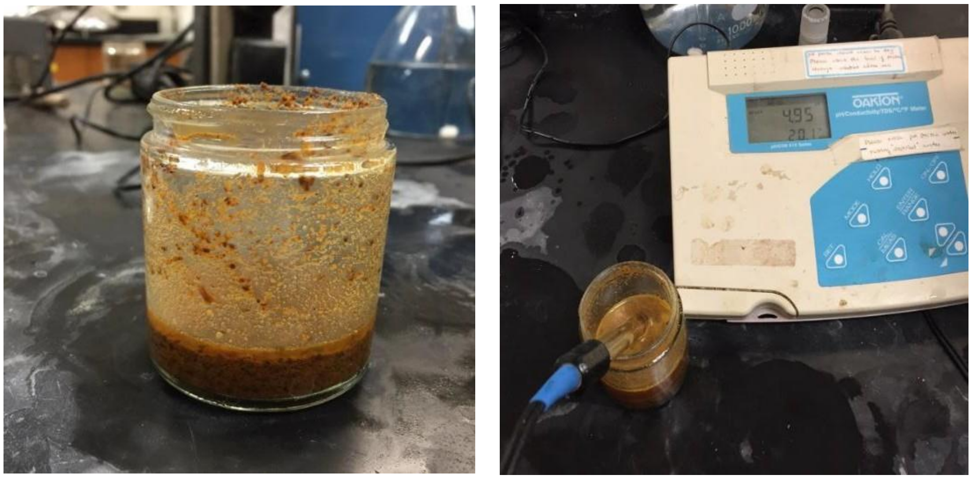


Figure 3.15 Determination of pH of dehydrated food waste

3.6.4 Carbon to Nitrogen (C/N) Ratio

Carbon to Nitrogen (C/N) ratio of the dehydrated food waste samples were determined using Perkin Elmer2400 CHNS/O Series II System. 10 mg of sample was measured and processed through the analyzer to determine the percentage of Carbon (C), Nitrogen (N) and Hydrogen (H) by mass (Figure 3.16). Subsequently, carbon and nitrogen percentages reported are used to determine the C/N ratio of the samples.

Figure 3.16 Determination of C/N ratio using CHN Analyzer

3.7 Tests on condensate water

The following properties of condensate water were evaluated to determine its potential usage as reclaimed water: pH, turbidity, Biochemical Oxygen Demand (BOD_5), enterococci and fecal coliform.

3.7.1 pH:

The pH of the condensate water was measured using bench top Oakton pH meter, as shown in Figure 3.17. The pH meter is calibrated using buffer solutions of known pH to ensure accurate measurement. After taking a reading, the pH probe was stored in a buffer solution (probe storage solution) to maintain a neutral pH.

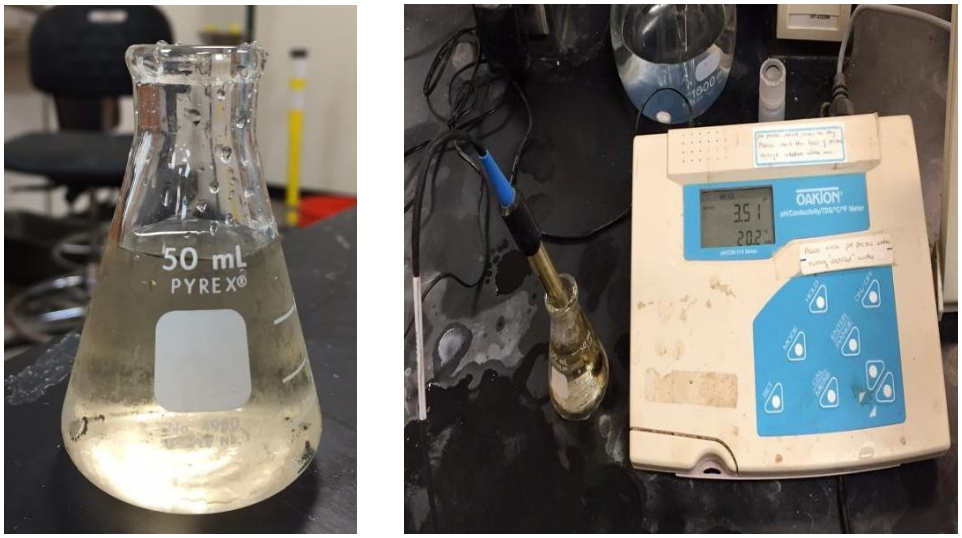


Figure 3.17 Determination of pH of Condensate Water

3.7.2 Turbidity

The turbidity of the condensate water was measured using Hach 2100P Portable turbidimeter (Figure 3.18). The turbidimeter was first calibrated using distilled water and solutions of known turbidity to ensure accurate readings.

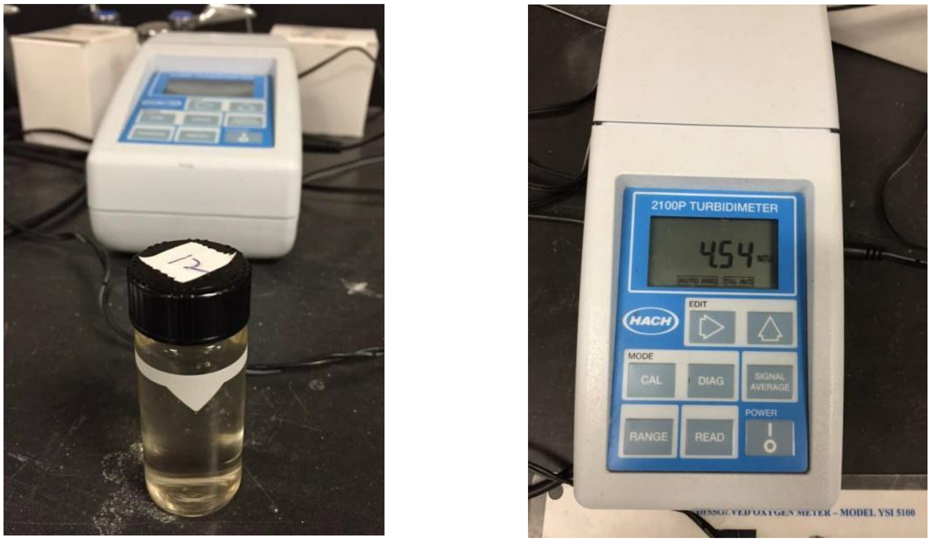


Figure 3.18 Measurement of turbidity of condensate water using Hach 2100P portable turbidimeter

3.7.3 Biochemical Oxygen Demand (BOD_5)

The biochemical oxygen demand (BOD_5) of the condensate water samples were determined following Standard BOD Procedure 8043. YSI 5100 benchtop dissolved oxygen instrument was used. Tests were performed at a dilution factor of 50 for institutional and household samples and at a dilution factor of 100 for restaurant and grocery samples, using seeded samples. Each test was performed in triplicates. The BOD probe was calibrated using distilled water samples and initial dissolved oxygen for each sample was measured. The samples were then capped and preserved in a 20^{o} C water bath for five days, after which the final dissolved oxygen was measured to calculate the biochemical oxygen demand of the samples (Figure 3.19).

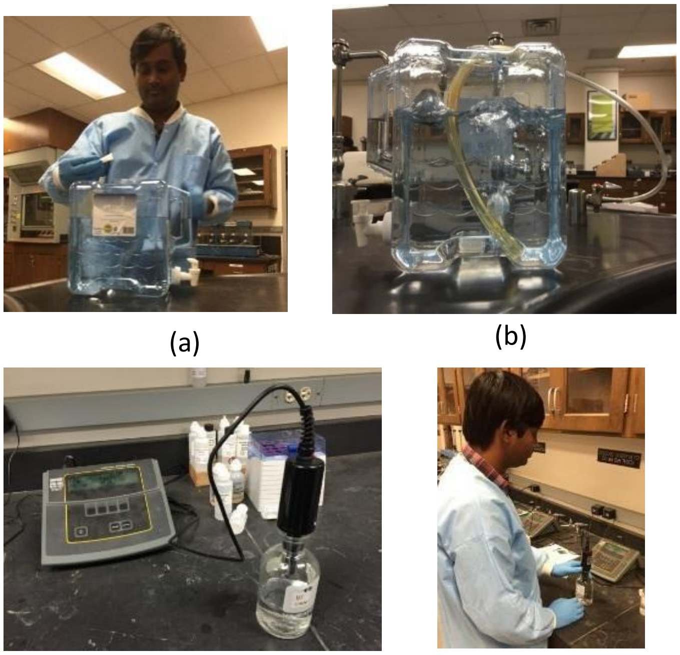
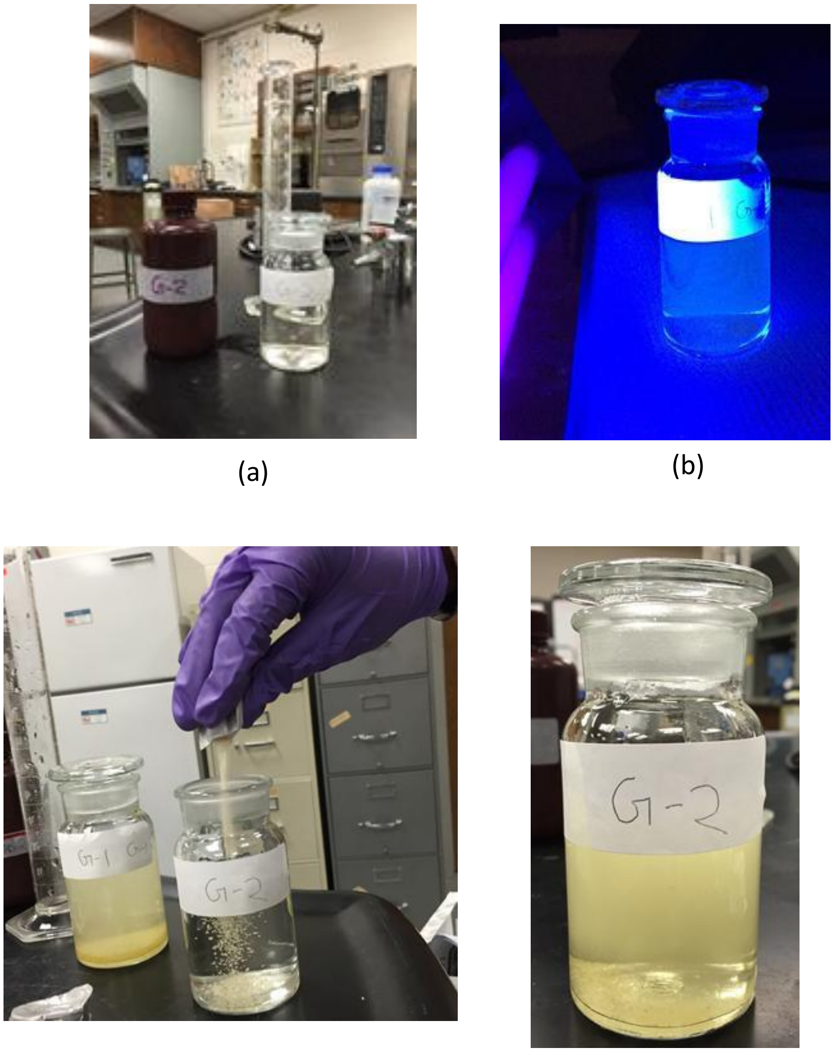

(a) (b)
(c) (d)

Figure 3.19 (a) Preparation of dilution water, (b) aeration of dilution water, (c) Dissolved oxygen probe calibration and (d) dissolved oxygen (DO) measurement

3.7.4 Determination of Enterococci Presence

Enterococci presence was determined using Enterolert® snap packs by IDEXX. To determine the presence of enterococci in condensate water samples, the contents of one snap pack of Enterolert ® was added to 100 mL of water sample in a sterile, non-fluorescing glass vessel. The powder was thoroughly mixed by shaking and the sample was incubated for 24 hours at 41^{o} C. After 24 hours, the samples were placed under UV light (365 nm) for fluorescence. Presence of fluorescence confirms the presence of enterococci in the water samples, whereas absence of fluorescence indicates absence of enterococci in the tested samples. Figure 3.20 shows the enterococci presence determination procedure.



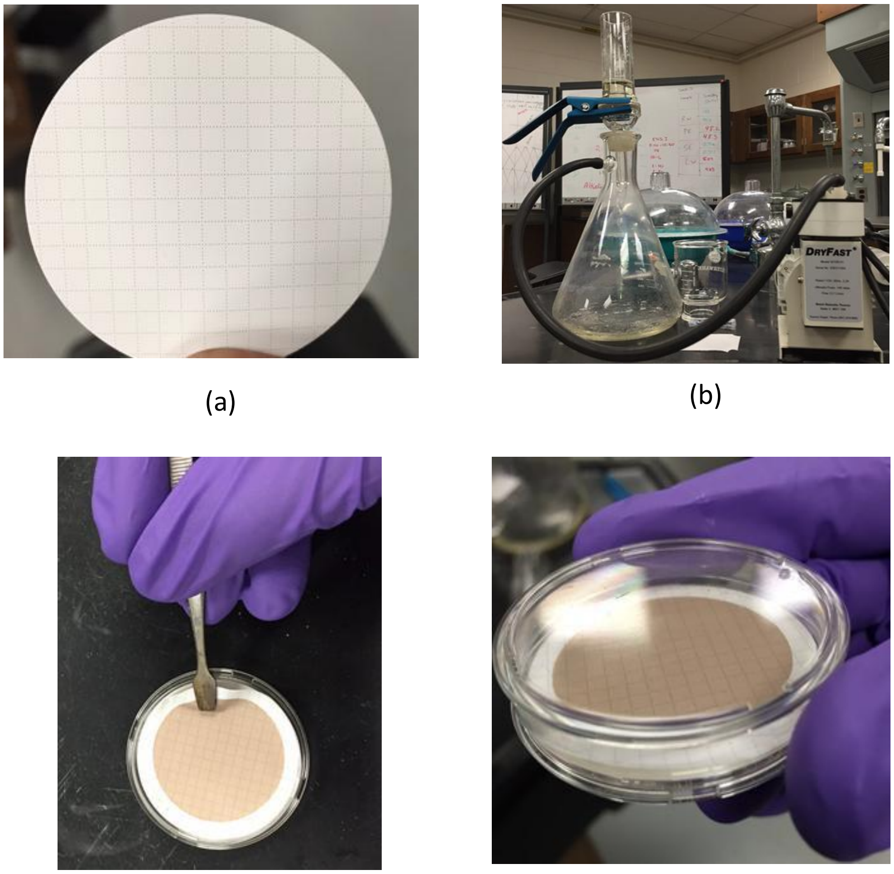
(a) (b)

(c) (d)

Figure 3.20 (a) 100 mL sample poured into sterile, non-fluorescing glass bottle, (b) Raw sample seen under UV light, (c) Adding a snap pack of Enterolert ® in the bottle, (d) Sample after addition of snap pack

3.7.5 Fecal Coliform Determination

Fecal Coliform was quantified using Membrane Filtration according to Standard Method 9222 D. First, dilution solution was prepared by making 0.8% saline solution (NaCl) and sterilizing the solution. A 47 mm petri dish with absorbent pads were soaked with 2 ml of fecal coliform (FC) medium. The excess medium was decanted from the petri dish. Then, 100 mL of sample was filtered through 0.45 μm gridded filter membrane with the help of a vacuum pump. The filter membrane was then transferred to the prepared petri dish aseptically with a pair of forceps. The petri dish was incubated at 44.5^{o} C for 24 hours to promote the growth of bacterial colonies. The fecal coliform colonies will appear in various shades of blue after 24 hours. Figure 3.21 shows the various steps of fecal coliform quantification.



(a) (b)

(c) (d)

Figure 3.21 (a) 0.45 μm gridded membrane filter, (b) filtration apparatus, (c) Placing membrane filter aseptically over absorbent pad in petri dish, (d) Sterile petri dish ready for incubation.

Chapter 4

Results and Discussions

4.1 Introduction

In this chapter, the results obtained by characterizing collected food waste from different sources, along with the physical, chemical and biological properties of the dehydrator outputs (dry FW and condensate water) will be presented and analyzed. The economics of dehydrator operation will also be discussed in this section to evaluate the sustainability and feasibility of food waste dehydrator operation.

Eight food waste samples from four different sources were collected and characterized for physical composition, moisture content and unit weight before processing them through the food waste dehydrator. The dehydrator outputs were collected and tested for their desired physical, chemical and biological properties to ascertain their potential sustainable usage. The dry FW samples were tested for their unit weight, particle size distribution, pH and C/N ratio to ascertain their suitability for compost feedstock. The condensate water samples were tested for their pH, turbidity, Biochemical Oxygen Demand (BOD_5), fecal coliform and enterococci presence to determine their utility as reclaimed water according to TCEQ regulations.

The results obtained in this study are discussed under four subsections: Physical properties of input waste, food waste dehydrator operation economics, properties of dry food waste (FW) and properties of condensate water.

4.2 Physical Properties of Food Waste

The following physical properties of input food waste were determined before operating the food waste dehydrator: physical composition, moisture content and unit weight.

4.2.1 Physical Composition of food waste

Two samples were collected from each of the four different sources considered. The collected food waste samples were manually sorted in five categories: Grains, Fruits and Vegetables, Meat, Seafood and Dairy products. The sorted individual components were weighed to determine their physical composition. Table 8 shows the physical composition of the food waste obtained from different sources.

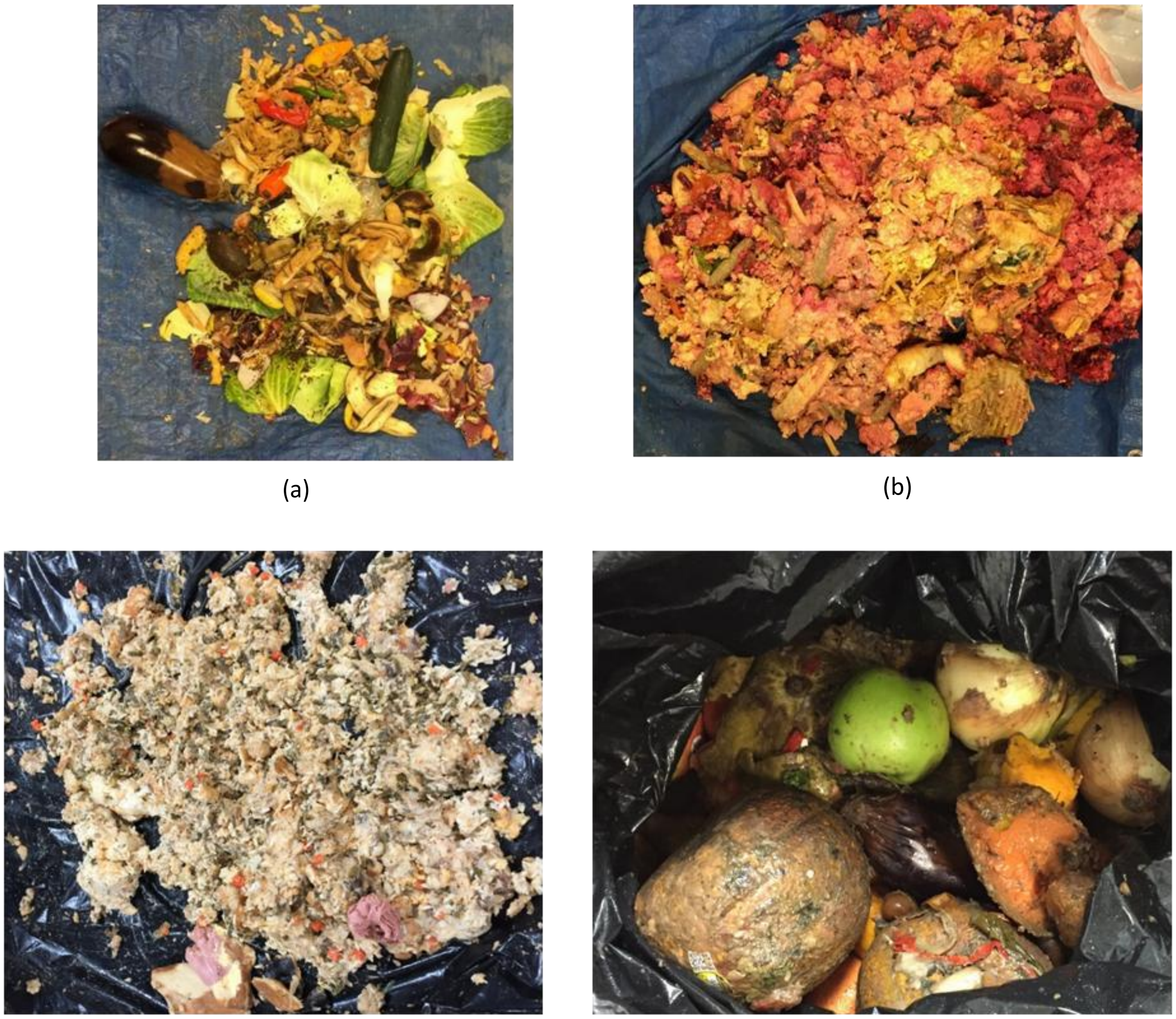
Table 8 Physical composition of food waste from different sources

Source	Sample No.	Physical Composition (% by Weight)				
		Fruits and Vegetables	Grain Products	Meat	Dairy	Seafood
Institutional	I-1	49.85	30.15	7.98	2.02	2
	I-2	29.45	21.05	30.10	0	0
Household	H-1	42.95	18.16	16.90	0.44	0.44
	H-2	63.90	8.20	14.70	1.90	1.9
Restaurant	R-1	16.04	74.88	5.70	0	0
	R-2	73.25	19.43	2.36	0	0
Grocery	G-1	100	0	0	0	0
	G-2	100	0	0	0	0

The food waste composition was generalized for each source by taking the weighted average of the sorted fraction from both the samples. The average composition of food waste from each source is shown in Figure 4.1.

Figure 4.1 Composition of food waste from different sources

Figure 4.1 indicates that fruits and vegetables constituted major fraction of the samples from all the sources except restaurant samples. The household samples had higher percentage of fruits and vegetables compared to institutional samples, since the household samples mostly consisted of vegetable peels and fruit discards. This observation is similar to the composition of household food waste reported by Banks et al. (2008), where segregated domestic food waste contained about 67% of cooked and uncooked fruits and vegetable. The percentage of meat reported (13%) in the study also aligns closely with the present study. However, the institutional and restaurant samples had higher percentage of grain products than household samples as the samples mainly contained uneaten pizza, breads and pasta. The data in the present study closely aligns with the study performed by Kim et al. (2006) on institutional food waste, where the food waste consisted of 45.3 ± 6.1% vegetables and 31.1 ± 3.3 % grains. Figure 4.2 shows a portion of samples from all sources. The other components (Dairy, meat and seafood) were similar for both the sources.



(a) (b)

(c) (d)

Figure 4.2 Fraction of food waste samples from (a) Household sources, (b) Institutional source, (c) Restaurant source and (d) Grocery source

Comparison with national average

ReFED (2016) provides the national food waste composition based on post-consumer food wastes. Comparison of compositions indicate that household and grocery wastes contain higher proportion of fruits and vegetables compared to the national average, while the percentage of fruits and vegetable in institutional waste closely aligns with the national average. However, an opposite trend is seen in case of grain products, where institutional and household samples have higher and lower percentage of food waste respectively when compared with the national average. Grocery, being a segregated source, contains only fruits and vegetables in its composition and is free of any other components. Restaurant waste has significantly higher grain products in its composition than national average and correspondingly lower proportion of fruits and vegetables. Also, all the sources have lower percentage of dairy compared to the national average. The physical composition of food waste from institutional and household sources, along with national composition is shown in Figure 4.3. It should be noted that while institutional, household and grocery samples were collected from source, restaurant samples were collected from landfill active face. Also, the national average are reported from mixed samples, while the samples in this study are source separated.

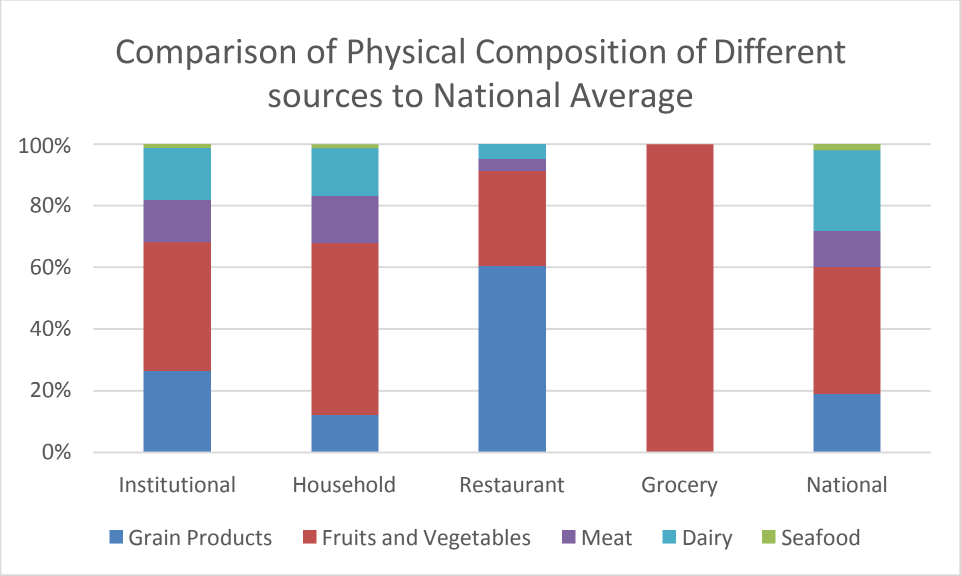


Figure 4.3 Comparison of physical composition of food wastes from different sources with national average

4.2.2 Moisture Content

The moisture contents of the samples from different sources are shown in Figure 4. 4.

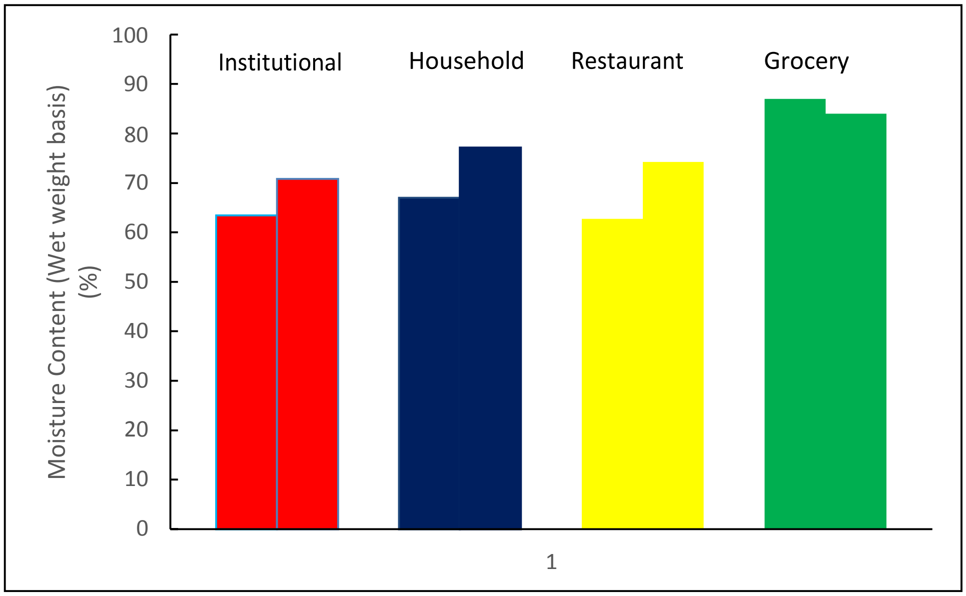


Figure 4.4 Moisture content of food waste from different sources

The moisture content of the institutional samples varied from 63-71%, household samples ranged between 67-77%, restaurant waste varied within 63-74%, whereas grocery waste samples fluctuated between 84-87%. The average moisture content of the institutional, household, restaurant and grocery waste was 67%, 72%, 68.5% and 85.5% respectively. Increased proportion of fruits and vegetables in food waste increases the moisture content of the sample (Kim et al., 2008). Hence, the household samples had more moisture content than the institutional samples as they had comparatively more fruits and vegetable content. This trend is further supported by very high (87%) moisture content of grocery samples, which consisted of only fruits and vegetables.

To ascertain the effect of fruits and vegetable content on the moisture content of the sample, variations of moisture content of the samples from all sources were plotted against their respective fruits and vegetable content as shown in Figure 4.5. Moisture content was observed to increase linearly with fruits and vegetable content ($R^2 = 0.80$).

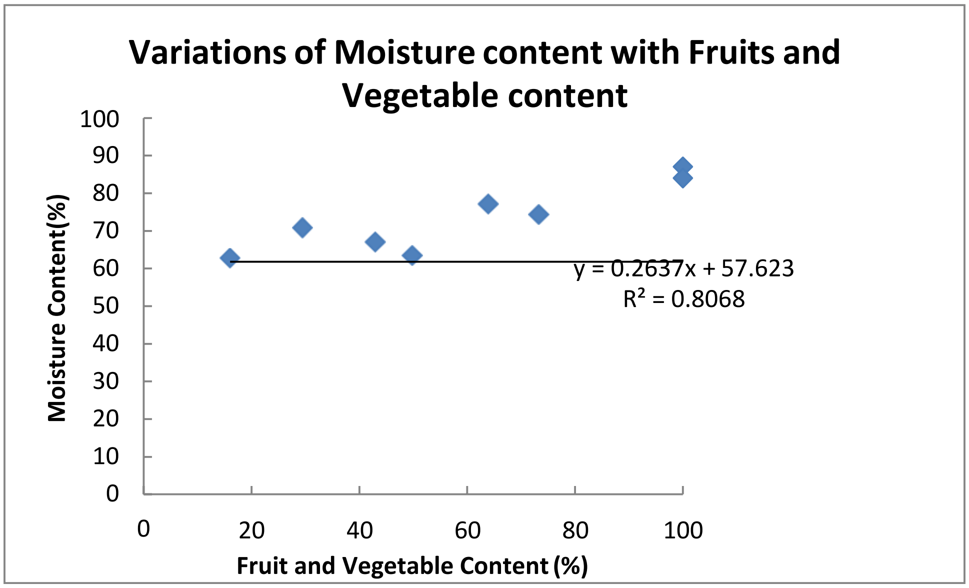


Figure 4.5 Variations of moisture content with fruits and vegetable content

Comparison with previous studies

The moisture content obtained from different sources is compared to the studies on food waste reported in the literature. The values of moisture content reported in the literature are summarized in Table 9.

The moisture content of the food waste reported in the literature varies from 70-85% for institutional waste, 65-80% for household waste and 90.5% for grocery waste. The average moisture content of the institutional samples in the current study (67%) is lower than the reported values. This may be attributed to the fact that most of the reported values in the literature are from Asian countries, which tend to have higher moisture content in their food waste. The moisture content reported in the literature for household and grocery wastes compares well with the values reported in the literature.

Table 9 Summary of moisture content of food waste found in literature

Source	Reference	Moisture content (%)	Location
Institutional	Present Study (2016)	67	Texas, USA
	Wang et al. (2014)	76	Zhejiang Gongshang University, China
	Brown and Murphy (2013)	70.6	University College Cork (UCC), Ireland
	Li et al. (2008)	85.3	Tongji University, China
	Zhang et al. (2012)	76.26	South Shropshire digestion facility, UK
	Chen et al. (2014)	73.1	Zhejiang University, China
	Kim and Shin (2008)	83.2	Korea Advanced Institute of Science and Technology, South Korea
	Kwon and Lee (2004)	80.03	University of Seoul, South Korea
	Dai et al. (2013)	79.6	Shanghai, China
	Shen et al. (2013)	77.39	University of Chemical Technology, Beijing, China

Household	Present study (2016)	72	Texas, USA
	Chang and Hsu (2008)	65–80	Kitchen waste, Taiwan
	Zhuang et al. (2008)	70.61	Hangzhou, China
Grocery	Present study (2016)	87	Texas, USA
	Shen et al. (2013)	90.5	Only fruit and vegetable waste from Beijing, China
	Jolanun&Towprayoon, (2010)	92.57	Only vegetable waste from Thailand

4.2.3 Unit weight

The unit weights of the samples from different sources are shown in Figure 4. 6.

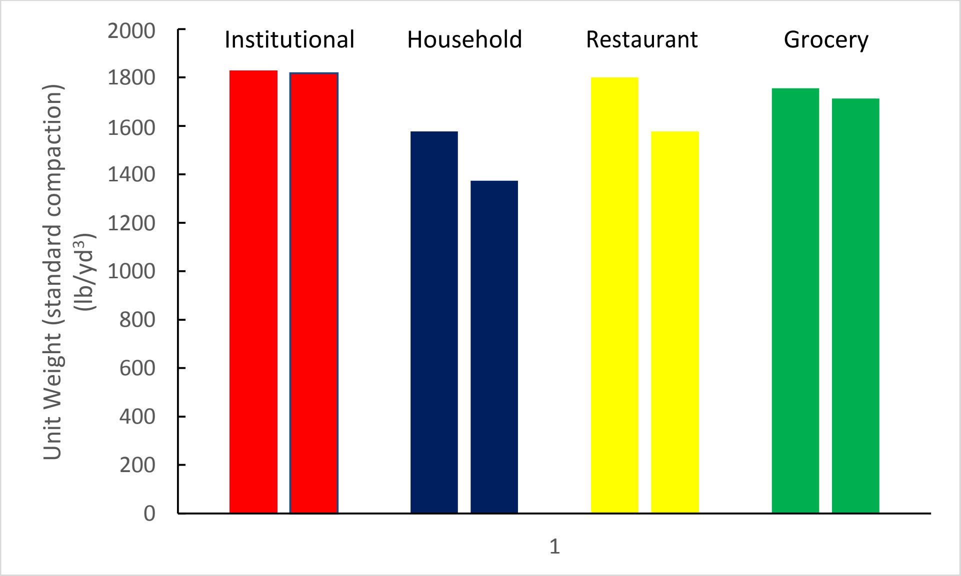


Figure 4.6 Unit weight of food waste samples from different sources

The unit weight of the institutional samples varied between 1819-1828 lb/yd^3, household samples ranged within 1373-1576 lb/yd^3, restaurant samples fluctuated within 1576.8 – 1800 lb/yd^3 and grocery samples varied from 1712.5 - 1755 lb/yd^3. Household wastes had the lowest unit weight, whereas institutional wastes had highest unit weight among the samples. The household wastes had relatively lower unit weight as it contained higher amount of bones, which were uncompressible. On the other hand, grocery samples had relatively high unit weight, as these samples contained uncontaminated fruits and vegetables which were very wet and readily compressible.

Comparison with previous studies:

The unit weights obtained from different sources is compared to the studies on food waste reported in the literature. The values of unit weights reported in the literature are summarized in Table 10.

Table 10 Summary of unit weight of food waste found in literature

Reference	Unit Weight kg/m^3(lb/yd^3)	Location and Remarks
Adhikari et al. (2009)	410 (691)	Food waste from composting centre (Canada)
Kim et al. (2008)	800 (1348.4)	Korea
EPA Victoria (2016)	1029 (1734.4)	Compacted kitchen waste (Australia)
Miller (2004)	1186.5 (2000)	Compacted landfilled food waste (USA)

The unit weight of the compacted food waste varied between 1734 lb/yd^3 to 2000 lb/yd^3. The average unit weights for the samples in this study were also aligned closely within this range, except for the household wastes which had lower unit weights.

4.3 Food waste dehydrator operation

GAIA food waste dehydrator was operated with the food waste obtained from different sources. The details of operation of the food waste dehydrator with different samples are reported in Table 11.

Table 11 GAIA food waste dehydrator operation details

Sample Type	Sample No.		Operation Details					
		Weight of input Sample (lb)	Weight of output food waste (lb)	**Weight loss (%)**	Volume of condensate water (Gallons)	Operation time (hrs)	Electricity Requirement (kWh)	**Cost of Operation (excluding facility charge) ($)**
Institutional	I-1	64.60	13.90	**78.50**	3.22	4.33	108.25	**2.50**
	I-2	42.89	13.19	**69.24**	2.05	3.33	83.25	**1.94**
Household	H-1	16.00	7.47	**53.00**	0.72	3.67	91.75	**2.13**
	H-2	24.82	9.55	**61.50**	1.50	3.33	83.25	**1.94**
Restaurant	R-1	37.28	7.74	**79.20**	2.30	3.33	83.25	**1.94**
	R-2	12.92	1.45	**88.77**	0.89	2.53	63.25	**1.47**
Grocery	G-1	59.26	10.5	**82.30**	4.90	3.70	92.50	**2.15**
	G-2	58.56	5.22	**91.08**	4.98	4.50	112.5	**2.62**

4.3.1 Weight loss

The total weights of the input and the output sample are shown in Table 12. The evaporation loss of the samples varied widely, from 11-58%. The institutional samples registered significantly higher evaporation loss than all the other samples. The evaporation losses of household and grocery samples were the lowest (11-15%).

The percentage of DFW and condensate water as output was also found to be widely varying. The grocery and restaurant samples had very high content of condensate water (71-88%), whereas the percentage of condensate water averaged between 50- 61% for institutional and household samples. The higher proportion of DFW in institutional and household wastes can be explained by their composition, since these samples contained a considerable proportion of animal bones in their composition, which were not effectively processed by the dehydrator as shown in Figure 4.7.

Table 12 Weight of input and output from food waste dehydrator

Sample Source	Sample No.	Weight of input sample (lb)	Total weight of output (DFW + condensate water) (lb)	Evaporation loss (lb)	Evaporation loss (% of input) (%)	DFW (% of output) (%)	Condensate water (% of output) (%)
Institutional	I-1	64.6	40.78	23.82	58.41	34.08	65.92
	I-2	42.89	30.30	12.60	41.60	43.50	56.50
Household	H-1	16.00	13.50	2.50	15.63	55.30	44.70
	H-2	24.82	22.08	2.74	11.04	43.25	56.75
Restaurant	R-1	37.28	26.95	10.33	27.70	28.72	71.28
	R-2	12.92	8.91	4.01	45.00	16.27	83.73
Grocery	G-1	59.26	51.42	7.84	13.23	20.42	79.60
	G-2	58.56	46.80	11.76	20.08	11.15	88.85

Figure 4.7 Unprocessed animal bones in the DFW fraction of institutional and household wastes

4.4 Properties of Dry Food Waste

4.4.1 Unit weight

The unit weight of the dry food waste sample was measured according to the Standard Proctor Compaction method (ASTM D698). The unit weight of the samples was determined for two different conditions. First, the mold was filled with the dry food waste samples without any compaction effort and the uncompacted unit weight was determined. The mold was then emptied and refilled in three layers with standard compaction effort. Both the uncompacted and compacted unit weights were determined to estimate the applicability of dry food waste for composting and landfilling respectively, since compaction in compost beds deters aerobic degradation and may lead to methane and nitrous oxide emission (Czepiel et al., 1996; Hellmann et al., 1997). The unit weights of the samples from different sources are reported in Figure 4.8.

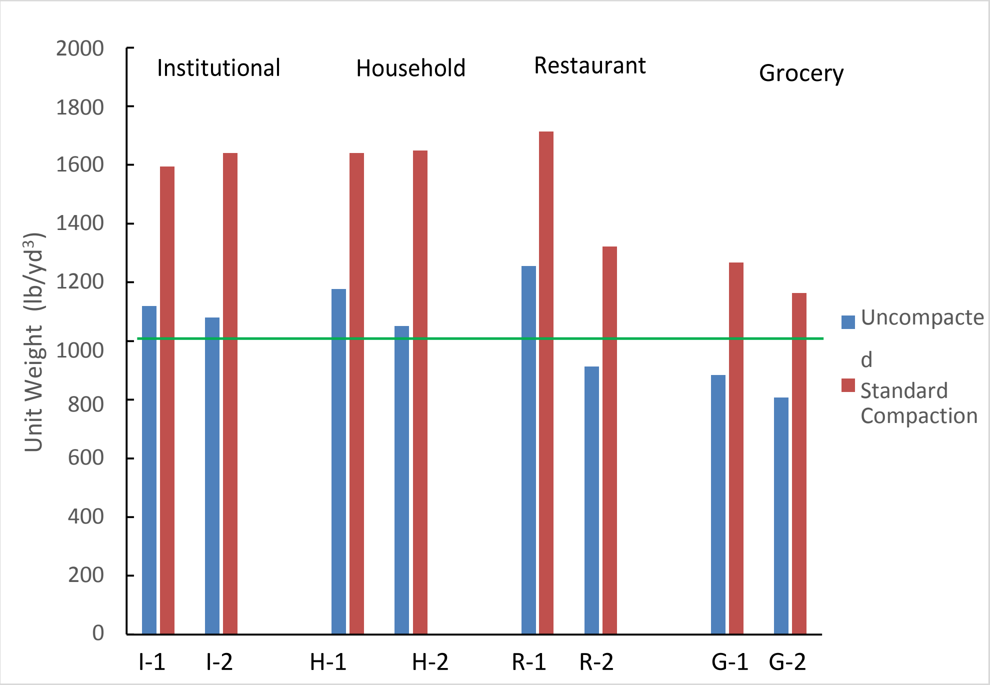


Figure 4.8 Unit weights of the dry Food Waste samples

According to Rynk (1992), the ideal unit weight of composting feedstock is about 1000 lbs/yd^3. From Figure 4.8, it is observed that all the food waste samples have uncompacted unit weight greater than 1000 lb/yd^3 except for grocery samples and restaurant sample R-1, which have slightly lower unit weights (806.5 – 912.5 lb/yd^3).The unit weight of the samples range from 806.5 lb/yd^3 to 1256.3 lb/yd^3 and are within reasonable range of the ideal value suggested. The range of unit weights obtained implies that dry food waste has the potential to be used as an ideal composting feedstock.

The compacted unit weight of the samples varied from 1267.1 – 1713.7 lb/yd^3. Dry FW shows high level of compaction when compared with average unit weight of fresh municipal solid waste, 35.85 pcf (968 lb/yd^3) (Taufiq, 2010). Higher unit weight at standard compaction indicates that the dry food waste samples are highly compressible and, if disposed at the landfill, can be helpful to achieve better compaction.

4.4.2 Particle Size Distribution

Dry food waste obtained by operating the food waste dehydrator was placed onto a stack of sieves arranged in descending order. The sieve assembly was shaken by a mechanical shaker to determine the particle size distribution of the dry food waste samples obtained from different sources. The particle size distribution of the samples is shown in Figure 4.9.

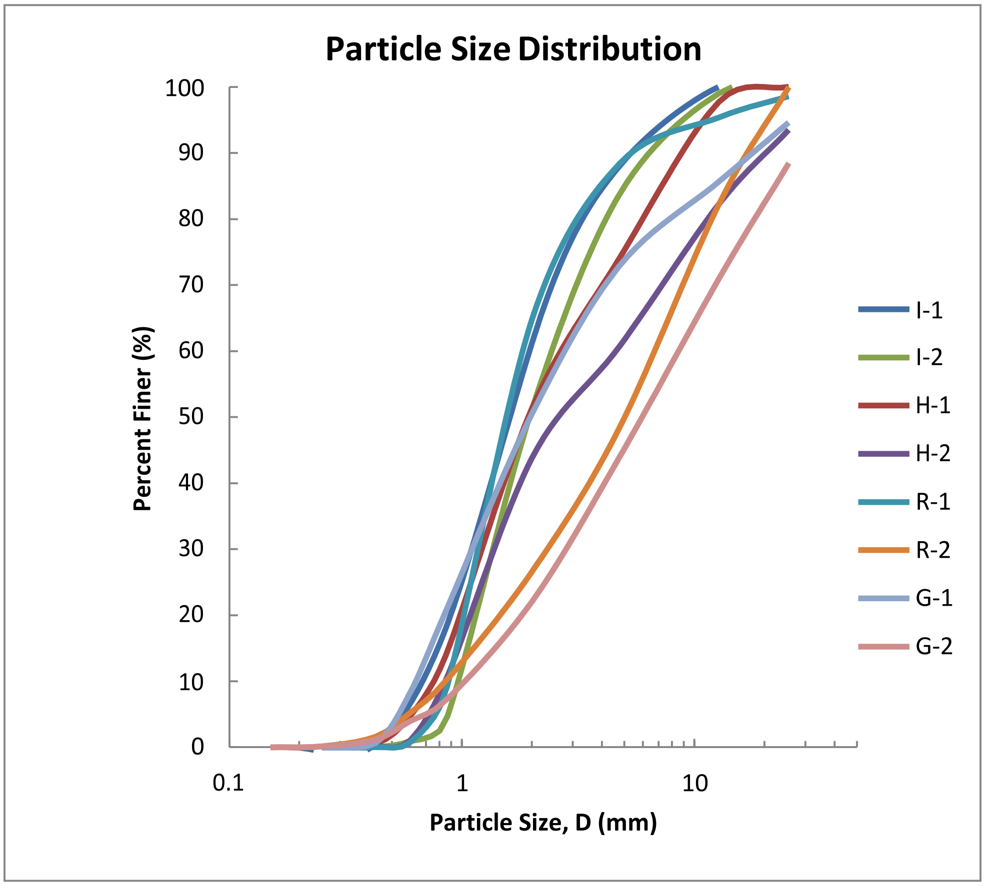


Figure 4.9 Particle size distribution of the dry food waste obtained from different sources

Figure 4.9 shows that the particle sizes of dry food waste obtained from different sources vary within a wide range, which implies that different component of waste stream are handled differently by the dehydrator. Household sample H-2 had animal bones as the major component of meat in its composition, which were retained on the 1'' sieve (Figure 4.7). Also, grocery samples had corn cobs and citrus fruit peels, which were also not effectively processed and were retained on the 1'' sieve (Figure 4.10).


(a) (b)

Figure 4.10 Portion of grocery samples retained on 1'' sieve (a) Corn cobs, (b) fruit peels

Among all the samples, the institutional samples were the finest and all the particles were finer than ½''. This can be explained by the absence of animal bones and whole fruits in its composition. On the other hand, grocery samples were much coarser, as they solely consisted of whole fruits and vegetables and the unprocessed fruit skins were retained on larger sieves. However, majority of the particle from all the sources were below 1'' and were highly variable in their sizes (0.15 mm – 25.4 mm). Variable size of particles provides greater surface area for microbial activity and is an indicator for good composting feedstock (Rynk, 1992). Hence, dry food waste obtained from dehydrator can serve as an ideal composting feedstock.

4.4.3 pH

pH of dehydrated food waste was determined by reading the pH of a 1:5 (Dry FW: Distilled water) solution, according to US Composting Council (USCC) recommendation. The pH of dry food waste samples obtained from different sources are summarized in Figure 4.11.

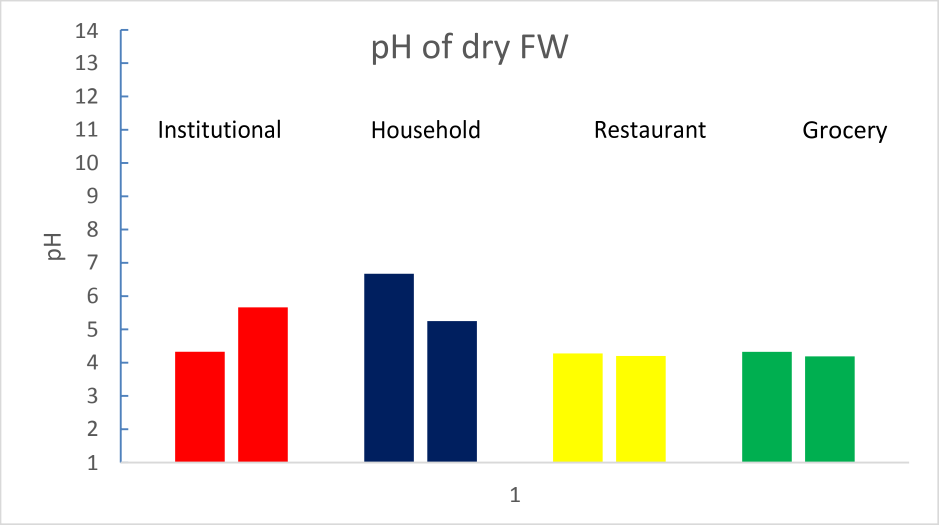


Figure 4.11 pH of Dehydrated Food Waste

The pH of dehydrated food waste (1:5 dilution) is found to be acidic for samples from all sources. The household samples had higher pH than all other sources, whereas the restaurant and grocery samples had much lower values of pH. The lower pH of grocery samples can be explained by the presence of citrus fruits and vegetables only, which were mainly acidic in nature. The pH of the samples were mostly acceptable for usage as a composting feedstock, indicating a sustainable usage of the obtained dry FW.

Comparison with previous studies

All of the studies reported in the literature dealt with fresh food waste rather than dehydrated food waste. However, their trend should be the same as dehydrated food waste is essentially food waste. The pH of food waste obtained from different sources is compared to the studies on food waste reported in the literature. The values of pH reported in the literature are summarized in Table 13.

The pH of the food waste reported in the literature is found only for institutional wastes. The pH reported varies from 4-6.1 for institutional food waste. The variation of pH for the institutional samples in the current study (5 ± 0.6) falls within the reported range. Additionally, the pH for all the samples within the current study (4.19-5.66) falls within the range specified in the literature, except for one sample from the household source (6.67).

Table 13 Summary of pH of food waste found in literature

Source	Reference	pH	Location
Institutional	Present Study (2016)	5 ± 0.6	Texas, USA
	Wang et al. (2014)	6.1	Zhejiang Gongshang University, China
	Brown and Murphy (2013)	4.1	University College Cork (UCC), Ireland
	Zhang et al. (2012)	4.71	South Shropshire digestion facility, UK
	Chen et al. (2014)	4.51	Zhejiang University, China
	Kim and Shin (2008)	4.6	Korea Advanced Institute of Science and Technology, South Korea
	Kwon and Lee (2004)	5.12	University of Seoul, South Korea
	Dai et al. (2013)	4.7 ± 0.7	Shanghai, China

4.4.4 Carbon to Nitrogen (C/N) Ratio

The Carbon to Nitrogen (C/N) ratio obtained by operating CHN analyzer with different samples are reported in Table 14.

Table 14 C/N Ratio of dry food waste samples

Sample	**C/N ratio**
I-1	12.7
I-2	12.4
H-1	14.4
H-2	14.0
R-1	15.3
R-2	7.4
G-1	**21.2**
G-2	**21.9**

From Table 14, it can be seen that the C/N ratio of the samples from the same source are similar to each other, except for the restaurant samples. The reason for the anomaly can be attributed to the fact that restaurant samples were collected from the active face of the City of Denton landfill at two different weeks. Consequently, they had a significant variation in their composition, leading to a variation in their C/N ratio also.

According to Rynk (1992), the C/N ratio of an ideal feedstock should be 20 – 40. From the results obtained in this study, it can be seen that all the samples except grocery samples has lower C/N ratio than recommended value. Hence, grocery samples are the most suitable samples for composting from the C/N ratio consideration.

4.5 Properties of Condensate Water

4.5.1 pH

The pH of the condensate water collected from the food waste dehydrator operation was measured using Oakton Benchtop pH meter. The pH of the condensate water obtained from different sources are summarized in Figure 4.12.

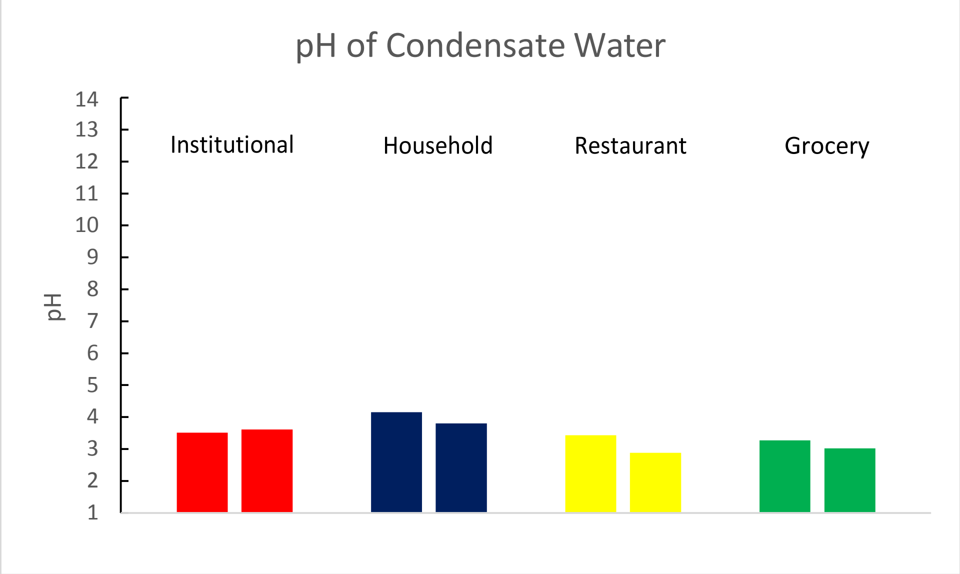


Figure 4.12 pH of condensate water obtained from different sources

From Figure 4.12, it can be seen that the condensate water obtained from all sources are acidic in nature. The household samples had the highest pH (3.51-3.61), whereas the restaurant and grocery samples had the lowest pH values. It is of interest to note that the pH of condensate water are lower than their solid counterparts and shows the same trend in variation. The higher values of pH in dry FW samples was due to dilution of the samples with de-ionized water (pH 7).

4.5.2 Turbidity

Turbidity of the condensate water samples from different sources were determined using Hach 2100P portable turbidimeter. Turbidity of condensate water is a specified parameter in Texas Commission on Environmental Quality (TCEQ) regulations for ascertaining suitability of discharged water for usage as reclaimed water. TCEQ requires that any discharged water should have turbidity below 3 Nephalometric Turbidity Unit (NTU) for Type-I reclaimed water usage (any type of usage where contact with public is possible). However, there is no limit for Type-II usage (usage where contact with public is improbable). The turbidity of condensate water obtained from dehydrator operation of various sources of food waste is shown in Figure 4.13.

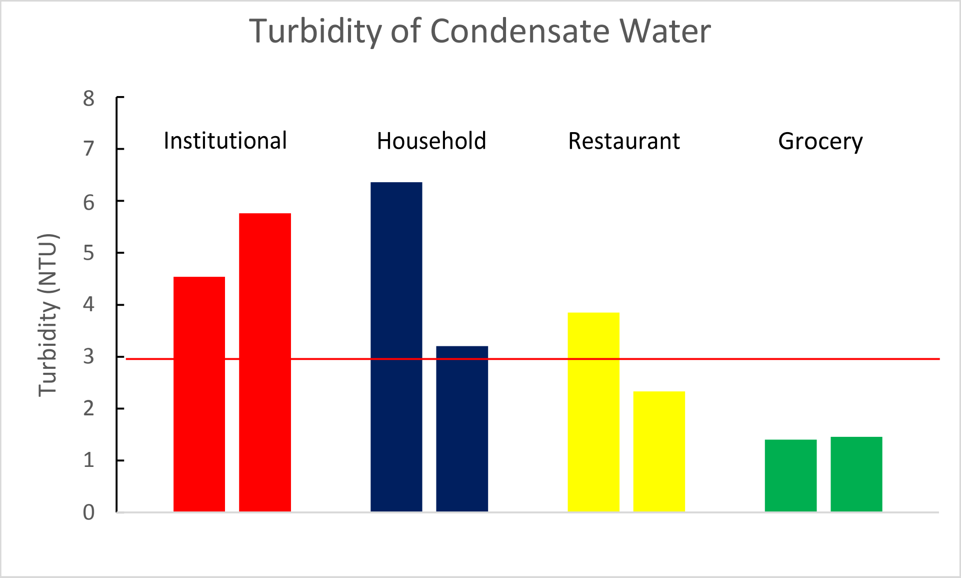
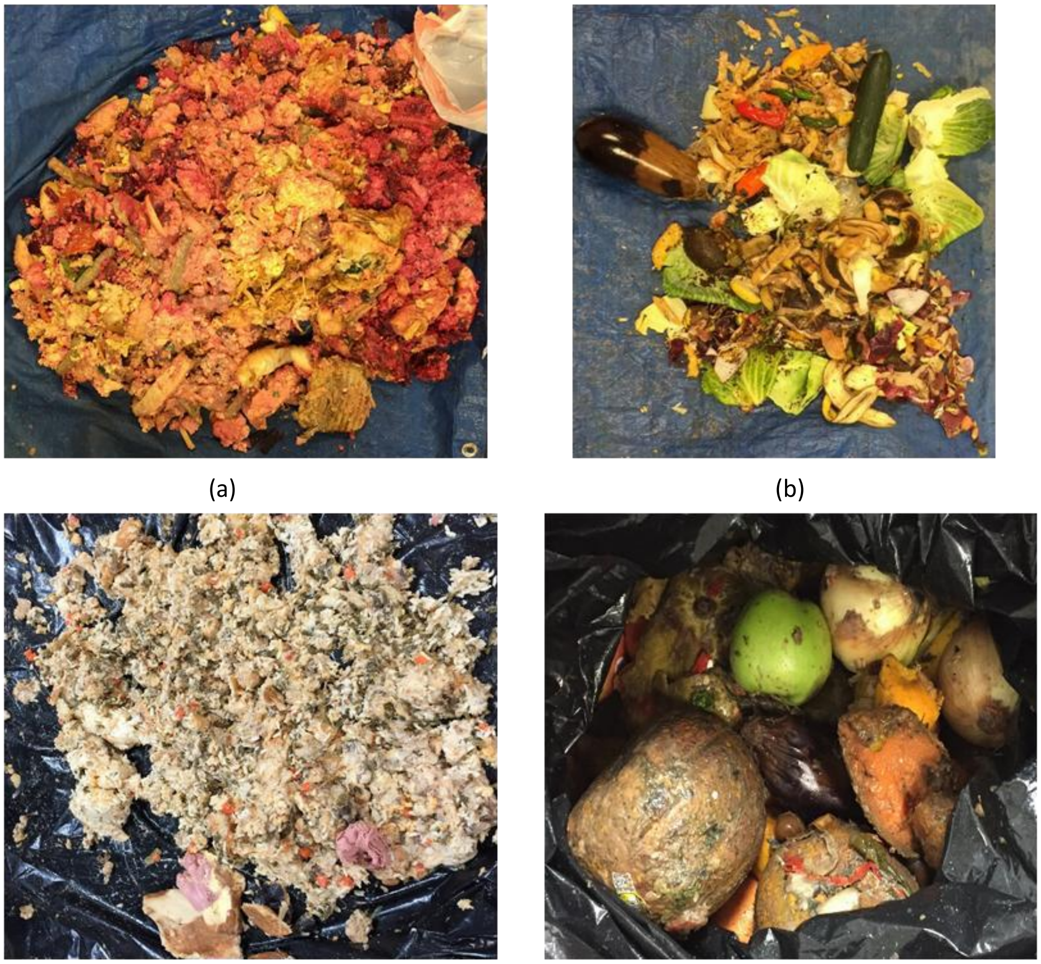


Figure 4.13 Turbidity of condensate water samples obtained from different sources

From Figure 4.13, it is observed that the institutional and household samples had higher turbidity than the restaurant and grocery samples. Also, there was a lot of variation within the household samples. However, the grocery samples had consistently low turbidity than all the other sources. The variation and trend in the turbidity can be explained from the physical composition of the samples. Both the institutional samples (I- 1 and I-2) were obtained from Connection Café in UT Arlington and mostly consisted of uneaten food cooked in oil (Figure 4.14). The same observation is true for the first household sample (H-1), which consisted mostly of food cooked in oil. However, the second household (H-2) sample mostly had uncooked fruits and vegetable discards in it and had much lower turbidity than the first one. Additionally, the first restaurant sample (R-1) was also found to consist mostly of cooked food (pasta). However, both the grocery samples (G-1 and G-2) had only fresh fruits and vegetables in their composition and consistently produced condensate water of low turbidity (Figure 4.14). This observation is further supported by Gammill et al. (2010) as they pointed out different components of oil contributed to increased turbidity. Hence, it can be concluded that presence of higher proportion of cooked food in food waste stream leads to higher turbidity in condensate water.



(a) (b)

(c) (d)

Figure 4.14 (a) Institutional Sample I-2, (b) Household Sample H-2, (c) Restaurant Sample R-1 and (d) Grocery Sample G-2

4.5.3 Biochemical Oxygen Demand (BOD_5)

Biochemical Oxygen Demand (BOD_5) of the condensate water samples from different sources were determined by following Standard BOD Procedure 8043. BOD5 of condensate water is a required parameter in Texas Commission on Environmental Quality (TCEQ) regulations to ensure use of discharged water as reclaimed water. TCEQ regulations mandate the BOD_5 of water for Type-I and Type-II reclaimed water usage must be less than 5 mg/l and 20 mg/l respectively. The BOD5 of condensate water obtained from dehydrator operation of various sources of food waste is shown in Figure 4.15.

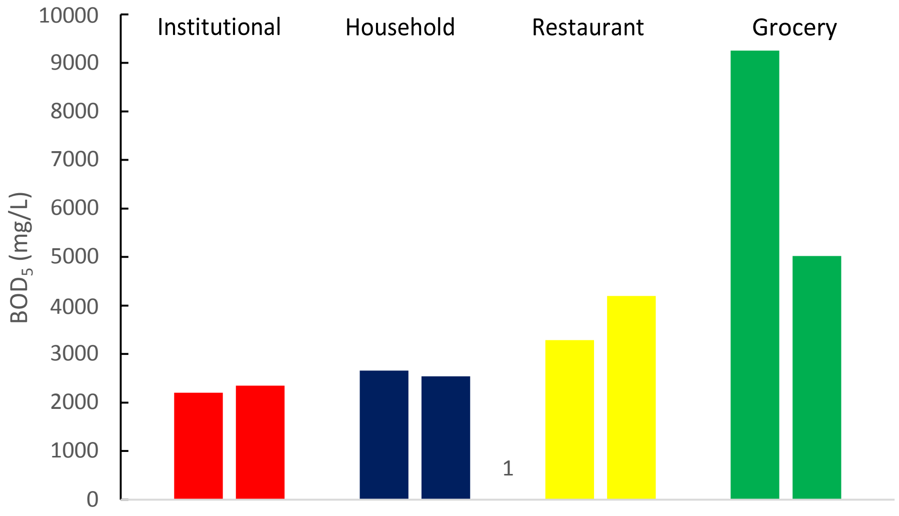


Figure 4.15 BOD_5 of condensate water samples from various sources

As shown in Figure 4.15, the BOD_5 of the condensate water samples from all sources exceeds the TCEQ requirements (5-20 mg/L) by a wide margin. Institutional and household samples had lower BOD_5 compared to the other two sources (2200-2660 mg/L). Restaurant samples had slightly higher BOD_5 values (3286-4196.7 mg/L). The grocery samples had the highest BOD_5 and also the largest variation between them

(5022-9250 mg/L). The BOD_5 of all the samples are excessively high and require appropriate treatment for BOD_5 removal before they can be used as reclaimed water.

4.5.5 Enterococci

The condensate water samples were tested for the presence of enterococci contamination. Enterococci quantification is one of the four TCEQ parameters for using discharge water as reclaimed water. TCEQ regulations state that the enterococci contamination should be limited to 4 Colony Forming Units (CFU)/100 ml for Type-I usage and 35 CFU/100 ml for Type-II usage. The samples were mixed with appropriate reagent and incubated for 24 hours before placing them under UV light and checking for fluorescence. Table 15 summarizes the findings for the Enterococci testing performed.

Table 15 Enterococci Presence/Absence Determination

Source	**Sample No.**	**Fluorescence**	**Enterococci Presence**
Institutional	**I-1**	**No**	**No**
	I-2	**No**	**No**
Household	**H-1**	**No**	**No**
	H-2	**No**	**No**
F **Restauran**r**t**	**R-1**	**No**	**No**
	R-2	**No**	**No**
o **Grocery** m	**G-1**	**No**	**No**
	G-2	**No**	**No**

From Table 15, it is observed that none of the samples had the presence of enterococci in them. Therefore, no microbiological treatment for enterococci removal is required to use the condensate water as reclaimed water. Figure 4.16 shows the absence of fluorescence in a condensate water sample (G-2).

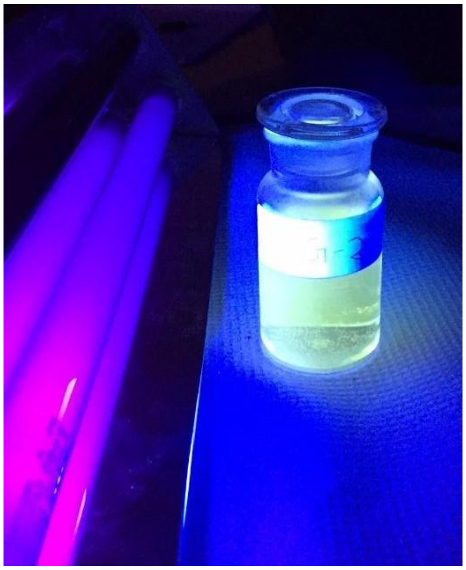


Figure 4.16 Non-fluorescence of sample G-2 indicating absence of enterococci in the sample

4.5.6 Fecal Coliform

The condensate water samples were also tested to quantify the amount of fecal coliform in the samples. Quantification of fecal coliform is a TCEQ parameter for using discharge water as reclaimed water. TCEQ regulations state that the fecal coliform contamination should be limited to 20 Colony Forming Units (CFU)/100 ml for Type-I usage and 200 CFU/100 ml for Type-II usage. 100 ml of sample was filtered on a filter paper and incubated on petri dish absorbent pad with fecal coliform medium at 44.5^{o} C for 24 hours. After 24 hours, the filtration pads were visually scanned to identify blue shaded fecal coliform colonies. Table 16 summarizes the findings for the fecal coliform testing performed.

Table 16 Fecal coliform colony determination

Source	Sample No.	Fecal coliform colonies
Institutional	**I-1**	**Not detected**
	I-2	**Not detected**
Household	**H-1**	**Not detected**
	H-2	**Not detected**
Restaurant	**R-1**	**Not detected**
	R-2	**Not detected**
Grocery	**G-1**	**Not detected**
	G-2	**Not detected**

From Table 16, it is seen that none of the samples had the presence of fecal coliform in them. Therefore, no microbiological treatment for fecal colifrom removal is required to use the condensate water as reclaimed water. Figure 4.17 shows the absence of CFU in a condensate water sample (G-2).


(a)
(b)

Figure 4.17 Petri dish with filtered sample (a) before incubation, (b) after incubation

Chapter 5

Potential Benefits of Dehydrator Operation

5.1 Weight Loss due to Dehydrator Operation

The dehydrator operation has been shown to reduce the incoming food waste significantly. Though the dehydrator efficiency was different for individual waste streams, all of the sources exceeded a weight loss of over 50%. Figure 5.1 shows the weight loss percentage achieved by food waste samples from different sources. From Figure 5.1, it is seen that grocery samples were processed by dehydrator with highest weight loss (82-91%), whereas household samples had the lowest weight loss (53-61%). The main reason of lower processing efficiency is attributed to the higher presence of animal bones in household samples, which were not adequately processed by the dehydrator.

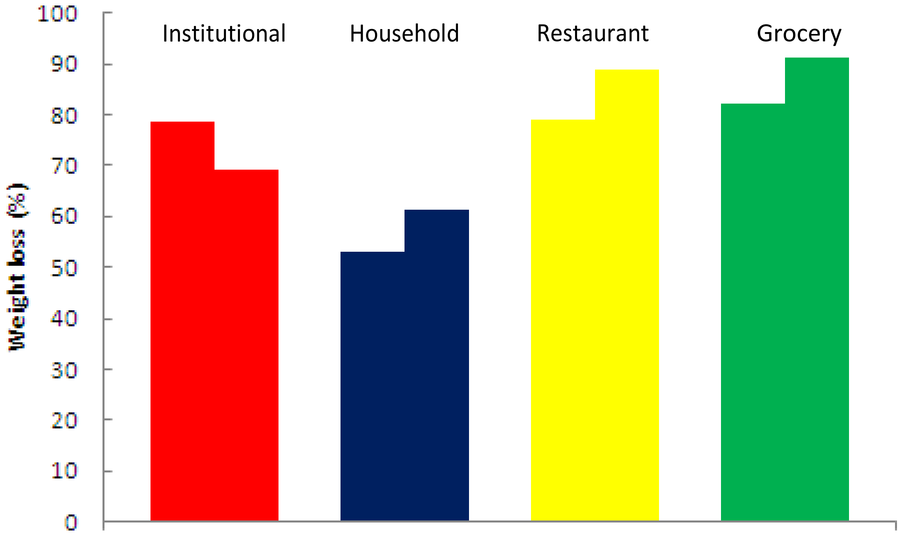


Figure 5.1 Weight loss percentage of food waste samples from different sources

5.2 Effect of weight loss on dehydrator economics

A simple calculation was carried out to evaluate the effect of weight loss through dehydrator operation on the operational economics of the dehydrator. While performing the calculations, the worst case scenario of the operation (dry FW has no further value or usage and will be disposed off in the landfill) was assumed to be on the conservative side. The weight loss was evaluated as the combination of weight of condensate water and evaporation loss as a percentage of input waste. The following calculations were performed to evaluate the weight loss economics.

Sample calculations:

Tipping fee (City of Denton) = $44/ton

Cost of airspace/lb. = $ (44/2000) = $ 0.022/lb.

Capacity of GAIA Food Waste Dehydrator = 200 kg = 440 lbs.

Maximum operation time of GAIA food waste dehydrator = 8.75 hours

Wattage of GAIA food waste dehydrator = 25 kW

Unit cost of electricity (City of Denton, commercial consumption) = $0.0233/kWh

Cost of Dehydrator operation = (8.75 hrs x 25 kW x $ 0.0233/ kWh)

= $ 5.10/cycle

Value of Airspace gained =$ [440*(X/100)*0.022],

where, X = Weight loss percentage

The weight loss percentage was plotted against the benefit-cost ratio (B/C) of operation, as shown in Figure 5.2.

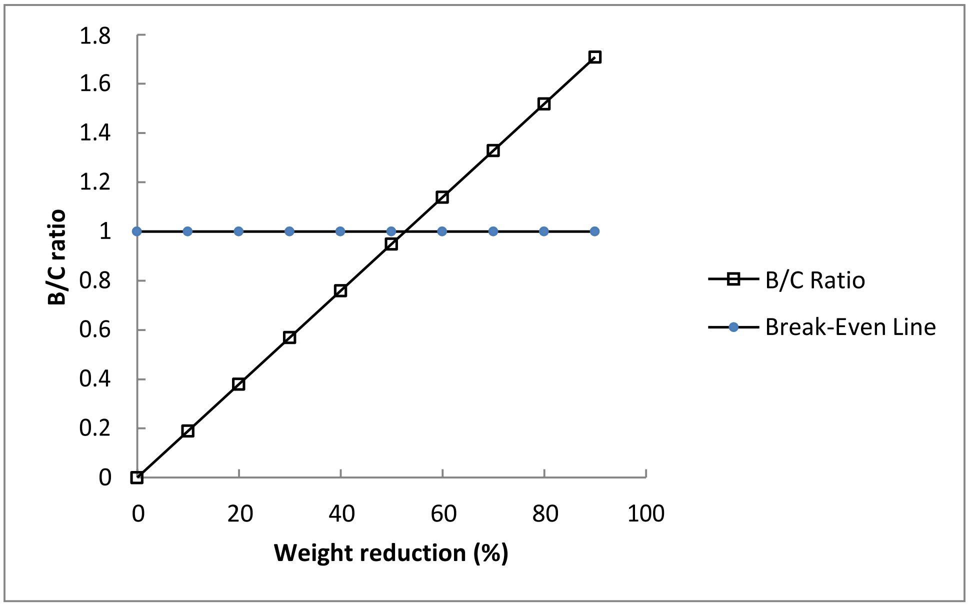


Figure 5.2 Economics of food waste dehydrator operation

From Figure 5.2, it is observed that the B/C ratio is less than 1 for weight loss percentage lower than 50%, indicating higher cost of operations than obtainable benefit in form of airspace gain. The break-even point is achieved at about 53%, where the cost of operation (per cycle) of GAIA food waste dehydrator equals the value of airspace gain. For weight reduction greater than 53%, the B/C ratio exceeds 1, denoting higher benefits than operating cost. Hence, the value of airspace gain is higher than the cost of operation for weight loss percentage greater than 53%, making the dehydrator operation economically feasible. The food waste samples from all the sources discussed in this study achieves a weight loss above the break-even point (except for the sample H-1, which equals the break-even point at 53% weight loss). The grocery samples achieved the highest weight loss (82-91%) and consequently, are most economically profitable to operate.

5.3 Benefits of dehydrator operation in landfill operations

To calculate the economic impact of food waste diversion using food waste dehydrator, the scenario of City of Denton was considered. Following are the assumptions and considerations undertaken for the economic analysis:

Daily MSW intake = 700 tons

Proportion of Food Waste = 2% (Taufiq 2010; Koganti, 2015)

Daily intake of Food Waste = 14 tons

Weight reduction due to dehydrator operation = 60% (conservative assumption)

The dehydrated food waste is landfilled (conservative assumption)

Following the above considerations and assumptions, the daily airspace gain and its corresponding value was calculated to understand the potential benefits of operating food waste dehydrator. The airspace gain and its value were calculated for different percentages of food waste diversion, as shown in Figure 5.3.

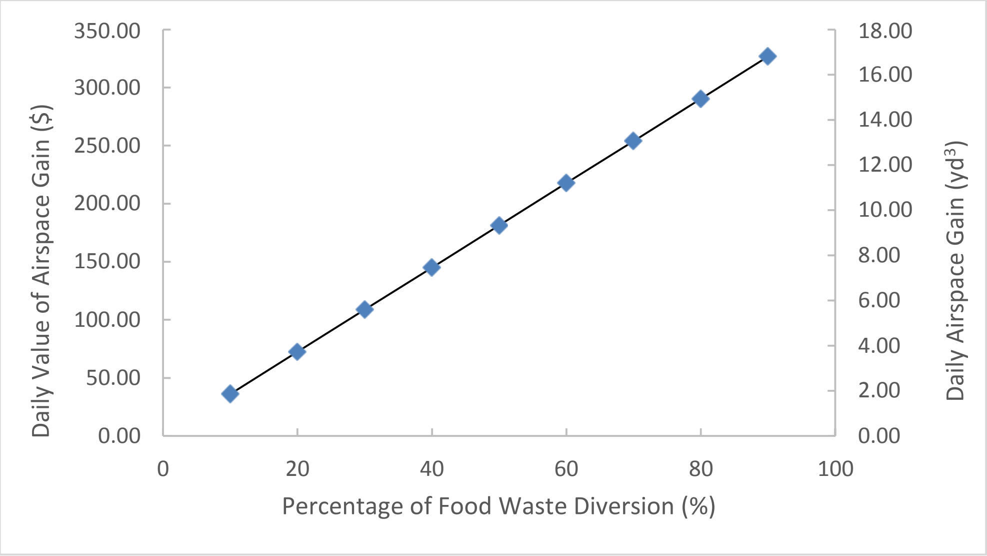


Figure 5.3 Benefits of food waste dehydrator operation for different diversion rates

From Figure 5.3, it can be seen that even with 50% diversion of daily incoming food waste to the food waste dehydrator, 9 yd^3 of airspace can be gained daily, valued at about $185. At this rate, City of Denton will be able to recover about 2700 yd^3 of airspace per year (considering 300 working days a year). However, the operating cost of the dehydrator has not been considered here, which will reduce the value of airspace gain. Nonetheless, from Figure 5.2, it can be inferred that for weight loss percentage of 60%, the benefits will outweigh the cost of operation.

5.4 Usage of dry food waste as compost feedstock

Rynk (1992) proposed a set of properties to identify suitable feedstock for composting operations. In this study, the dry food waste obtained by operating food waste dehydrator were tested for the following properties and checked against the recommended limits, as shown in Table 17.

Table 17 Acceptable properties of ideal feedstock for composting

Property	**This study**	**Proposed by Rynk (1992)**
Bulk Density (lb/yd^3)	806-1256	1000
Particle size	Mostly <1 in. and variable	< 1 in. and variable
pH (1:5 dilution)	4.19-6.67	5.5-9.0

From Table 17, it is evident that the tested samples from different sources mostly satisfy the range of desired properties of ideal feedstock. Hence, the dehydrated food waste can be successfully used as a composting feedstock to produce quality compost instead of being buried in the landfill.

5.5 Usage of condensate water as reclaimed water for landscaping

Texas Commission on Environmental Quality (TCEQ, 2003) puts forward a set of recommended physical, chemical and biological property limits to determine the suitability of treated water for reclaimed water usage. In this study, the condensate water samples obtained from dehydrator operation were investigated for the proposed limits to identify their suitability for landscaping usage, as shown in Table 18.

Table 18 Comparison of obtained parameters with TCEQ limits for reclaimed water usage

Property	This Study	TCEQ Limits	
		Type-I Usage*	Type-II Usage**
Turbidity (NTU)	1.4 – 7.15	3	-
BOD_5 (mg/l)	2200 - 9250	5	20
Fecal Coliform	Not detected	20 CFU/100 ml	200 CFU/100ml
Enterococci	Absent	4 CFU/100 ml	35 CFU/ 100ml

*Type I Usage includes uses where the public may come in contact with the reclaimed water

** Type II Usage includes uses where the public would not come in contact with the reclaimed water

Table 18 indicates that all the recommended limits of TCEQ are satisfied by condensate water samples for landscaping, except for turbidity in Type-I usage by some samples and exceedingly high BOD_5 for the other samples. It is of interest to note that presence of cooked food increases the turbidity in the samples. Hence, the condensate water samples can be used for landscaping usage with appropriate treatment for BOD_5 reduction. However, the BOD_5 of the samples are within the limit for discharge into the sewer and can be processed normally by waste water treatment plants.

5.6 Comparison of dehydrator operation with other on-site alternatives

On-site food waste treatment technologies can broadly be classified into two categories – Biological digesters and non-biological systems. Biological digesters include aerobic systems (in-vessel composters) and anaerobic digesters, performing wet or dry operations based on feedstock. Non-biological systems include pulper/shredder and dehydrators, which combine mechanical shredding and heat to achieve weight/volume reduction. The advantages and limitations of the various on-site treatment systems are compared in Table 18. From Table 19, it is evident that food waste dehydrators present many more benefits and is more economical and efficient to operate than the other on-site food waste treatment alternatives.

Table 19 End products, advantages and limitations of on-site food waste treatment systems

System	Accepted wastes	End products	Advantages	Limitations
Anerobic Digesters (AD)	All, except woody organics	i) Solid digestate ii) Water (nutrient rich) iii) Biogas (about 70% methane)	i) Digestate used as fertilizer ii) Power generation	i) High capital cost ii) High process control
Aerobic Systems (In-vessel composting)	Only selective food waste	Wet- Nutrient rich wastewater	i) Volume reduction	i) BOD of wastewater too high for treatment ii) Addition of micro-organisms
		Dry - Compost	i) Compost	i) Addition of micro-organisms ii) Unstable compost iii) Long processing time (1-14 days)
Dehydrators	All sorts of food waste, including soiled paper and napkins	Dry food waste and condensate water	i) Weight and volume reduction ii) Dry food waste as composting feedstock iii) No process control iv) Low capital cost v) Low processing time	i) High BOD wastewater

Chapter 6

Conclusions and Recommendations

The main objective of the study was to evaluate the diversion potential of food waste from the landfill using food waste dehydrators and recommend sustainable usage of the obtained outputs. To realize this objective, a total of 8 samples were collected from four different sources (Institutional, Household, Restaurant and Grocery) and their physical properties were determined. The samples were then processed by operating the food waste dehydrator and the outputs of operation (dry food waste and condensate water) were collected. The dry food waste was tested for its unit weight, particle size distribution and pH to ascertain its suitability as a composting feedstock, whereas the condensate water was tested for its pH, turbidity, BOD_5, enterococci and fecal coliform to determine its usability as reclaimed water. The results obtained in this study are summarized in the following subsection.

6.1 Summary and Conclusions

- A total of 8 samples, two from each source (institutional, household, restaurant and grocery) were collected. Fruits and vegetables and grain products were the major components in the food waste stream. Fruits and vegetables were found to be the dominant component in all the waste sources except for restaurant.
- The moisture content of the food wastes ranged from 63-87% for all the samples. Moisture content of the waste stream was also found to be directly proportional to the fruits and vegetable content in the waste

stream ($R^2 = 0.80$).The unit weight of the food waste samples ranged from 1373-1828 lb/yd^3.

- The weight loss percentage due to food waste dehydrator operation was found to vary from 53% - 91%. The weight loss for household samples were lowest (53-61%), whereas grocery samples registered highest weight loss (82%-91%).
- The uncompacted density of dry food waste was found to vary from 806 lb/yd^3 to 1256 lb/yd^3 and compacted density (standard compaction) ranged between 1163-1713 lb/yd^3. These densities were within reasonable range of optimum unit weight for compost feedstock, 1000 lb/yd^3.
- The particle size of the dry food waste also varied within a large range. The institutional samples were the finest and the grocery and household samples were found to be coarser. Presence of unprocessed animal bones and whole fruit skins contributed to coarser sizes of household and grocery samples respectively. Overall, the varied particle size of the sample will provide more surface area for microbial activity and makes dry food waste an ideal compost feedstock.
- The pH of dry food waste (1:5 dilution) indicated that all the samples were acidic in nature. The grocery and restaurant samples had lower pH values, whereas household and institutional samples had relatively higher pH values.
- The C/N ratio of most samples was found to be below 20. However, the grocery samples had the highest C/N ratio among all the samples.

- The condensate water sample followed the same trend as their solid counterparts (dry food waste), with grocery and restaurant samples being more acidic than the institutional and household samples. However, for the same source, the pH of the condensate water samples were lower than dry food waste samples.
- The turbidity of the samples was found to vary from 1.46 – 7.15 NTU. Grocery samples had the lowest turbidity, whereas institutional and household samples were found to have significantly high turbidity. Presence of cooked food in waste stream seemed to have an increasing effect on turbidity, as presence of oil contributes turbidity in the samples.
- The BOD_5 of condensate water samples ranged between 2200 mg/l to 9250 mg/l. The BOD_5 of water samples were found to be exceedingly higher than the proposed regulatory limit for reclaimed water usage (5-20 mg/l). Hence, the condensate water samples should be treated for BOD_5 removal before using them as reclaimed water.
- The condensate water samples from all sources were found to be free of enterococci and fecal coliform contamination, indicating that no biological treatment for bacteria removal is necessary to use them as reclaimed water.
- A weight reduction of 53% or more makes the food waste dehydrator operation economically feasible. All the samples considered in this study had weight loss percentage of 53% or higher, hence making food waste dehydrator operation economically feasible for all the sources considered.

- With 50% food waste diversion in City of Denton Landfill, 68 yd^3 of airspace can be gained daily, valued at $ 1346.
- The dry food waste meets the criteria for ideal feedstock and may be used in composting to produce quality products.
- The condensate water needs to be treated for BOD removal before being used as reclaimed water.
- Among the different alternatives for on-site food waste processing, food waste dehydrators are most efficient and economical to operate sustainably.

6.2 Recommendation for future study

Based on the limitations of the current study, following recommendations are proposed for future study:

- More samples of food waste may be collected to get representative data on food waste composition for different sources.
- Different volumes of food waste can be processed in the food waste dehydrator to study the effect of food waste weight on dehydrator performance.
- A full scale composting operation can be performed on the composting feedstock obtained only from the dehydrator to ascertain whether the compost with this feedstock results in a better quality compost.
- Life Cycle Analysis (LCA) can be performed on food waste dehydrator operation to ascertain the environmental impact of food waste dehydrator operation and compare it with other food waste treatment alternatives.

References

Abdullahi, Y. A., Akunna, J. C., White, N. A., Hallett, P. D., & Wheatley, R. (2008). Investigating the effects of anaerobic and aerobic post-treatment on quality and stability of organic fraction of municipal solid waste as soil amendment. *Bioresource technology*, *99*(18), 8631-8636.

Adhikari, B. K., Barrington, S., Martinez, J., & King, S. (2009). Effectiveness of three bulking agents for food waste composting. *Waste management*,*29*(1), 197-203.

AR News (January 2014). More states ban organic waste in landfills. http://www.americanrecycler.com/0114/2428more.shtml

Banks, C. J., & Stringfellow, A. (2008). A pilot-scale trial comparing mesophilic and thermophilic digestion for the stabilisation of source segregated kitchen waste.

Beck, R. W. (2001). US recycling economic information study. *Prepared for the National Recycling Coalition. Environmental Protection Agency*.

Bergstrom and Rasmussen (2011). *Food Waste Diversion at Urban University.* BioCycle. Vol. 52, No. 12, p. 34.

BioCycle (2013).http://www.biocycle.net/2013/10/25/analysis-of-biodigesters-and-dehydrators-to-manage-organics-on-site/

Browne, J. D., & Murphy, J. D. (2013). Assessment of the resource associated with biomethane from food waste. *Applied Energy*, *104*, 170-177.

Buzby, J. C., Hyman, J., Stewart, H., & Wells, H. F. (2011). The value of retail ‐and consumer ‐level fruit and vegetable losses in the United States. Journal of Consumer Affairs, 45(3), 492-515.

Cal Recycle (2014). Food scraps management technologies. http://www.calrecycle.ca.gov/Organics/Food/Commercial/Dehydrators.htm

Chang, J. I., & Hsu, T. E. (2008). Effects of compositions on food waste composting. *Bioresource Technology*, *99*(17), 8068-8074.

Chen, X., Yan, W., Sheng, K., & Sanati, M. (2014). Comparison of high-solids to liquid anaerobic co-digestion of food waste and green waste.*Bioresource technology*, *154*, 215-221.

Chen, Y., Xiao, B., Chang, J., Fu, Y., Lv, P., & Wang, X. (2009). Synthesis of biodiesel from waste cooking oil using immobilized lipase in fixed bed reactor. *Energy Conversion and Management*,*50*(3), 668-673.

Cook, P. R. M. (2015). An Analysis of New and Emerging Food Waste Recycling Technologies and Opportunities for Application. http://greatforest.com/wp-content/uploads/2015/08/New-and-emerging-food-waste-recycling-technologies-RC.pdf

Cuéllar, A. D., & Webber, M. E. (2010). Wasted food, wasted energy: The embedded energy in food waste in the United States. Environmental science & technology, 44(16), 6464-6469.

Czepiel, P., Douglas, E., Harriss, R., & Crill, P. (1996). Measurements of N2O from composted organic wastes. *Environmental Science & Technology*,*30*(8), 2519-2525.

Dai, X., Duan, N., Dong, B., & Dai, L. (2013). High-solids anaerobic co-digestion of sewage sludge and food waste in comparison with mono digestions: Stability and performance. *Waste management*, *33*(2), 308-316.

Ebel, R., Borchers, A., & Carriazo, F. (2011). Major uses of land in the United States, 2007. US Department of Agriculture, Economic Research Service.

Environmental Protection Agency (EPA) Victoria. Waste Materials – Density Data. Retrieved May 16, 2016, from http://www.epa.vic.gov.au/business-and-industry/lower-your-impact/~/media/Files/bus/EREP/docs/wastematerials-densities-data.pdf

FAO. (1981). Food Loss Prevention in Perishable Crops. Rome: Food and FAO Agricultural Service Bulletin, no. 43, FAO Statistics Division.

FAO. (2011). Global food losses and food waste – Extent, causes and prevention. Rome

Galanakis, C. M. (2012). Recovery of high added-value components from food wastes: conventional, emerging technologies and commercialized applications. Trends in Food Science & Technology, 26(2), 68-87.

Gammill, W., Proctor, A., & Jain, V. (2010). Comparative study of high-linoleic acid vegetable oils for the production of conjugated linoleic acid. *Journal of agricultural and food chemistry*, *58*(5), 2952-2957.

Goldman, G., & Ogishi, A. (2001). The Economic Impact of Waste Disposal and Diversion in California. University of California, Berkeley.

Gunaseelan, V. N. (2004). Biochemical methane potential of fruits and vegetable solid waste feedstocks. Biomass and bioenergy, 26(4), 389-399.

Gunders, D. (2012). Wasted: How America is losing up to 40 percent of its food from farm to fork to landfill. Natural Resources Defense Council.

Hartmann, H., & Ahring, B. K. (2006). Strategies for the anaerobic digestion of the organic fraction of municipal solid waste: an overview. Water science and technology, 53(8), 7-22.

He, M. X., Feng, H., Bai, F., Li, Y., Liu, X., & Zhang, Y. Z. (2009). Direct production of ethanol from raw sweet potato starch using genetically engineered Zymomonas mobilis. Afr J Microbiol Res, 3(11), 721-726.

He, M., Sun, Y., Zou, D., Yuan, H., Zhu, B., Li, X., & Pang, Y. (2012). Influence of temperature on hydrolysis acidification of food waste. Procedia Environmental Sciences, 16, 85-94.

Hellmann, B., Zelles, L., Palojarvi, A., & Bai, Q. (1997). Emission of Climate-Relevant Trace Gases and Succession of Microbial Communities during Open-Windrow Composting. *Applied and environmental microbiology*,*63*(3), 1011-1018.

ISWA (2013). Food Waste as a global issue. Key issue paper. http://waste.ccac-knowledge.net/document/iswa-key-issue-paper-food-waste-global-issue-perspective-municipal-solid-waste-management

Jolanun, B., & Towprayoon, S. (2010). Novel bulking agent from clay residue for food waste composting. *Bioresource technology*, *101*(12), 4484-4490.

Katami, T., Yasuhara, A., & Shibamoto, T. (2004). Formation of dioxins from incineration of foods found in domestic garbage. Environmental science & technology, 38(4), 1062-1065.

Kim, D. H., & Kim, M. S. (2013). Development of a novel three-stage fermentation system converting food waste to hydrogen and methane.Bioresource technology, 127, 267-274.

Kim, D. H., Kim, S. H., & Shin, H. S. (2009). Hydrogen fermentation of food waste without inoculum addition. Enzyme and Microbial Technology, 45(3), 181-187.

Kim, H. J., Kim, S. H., Choi, Y. G., Kim, G. D., & Chung, T. H. (2006). Effect of enzymatic pretreatment on acid fermentation of food waste. *Journal of Chemical Technology and Biotechnology*, *81*(6), 974-980.

Kim, H.Y. (2014). Decision-Making in the Selection of Food Waste Diversion Systems for Boone, North Carolina: Comparing Composting and Anaerobic Digestion by Life

Cycle Assessment and Cost Benefit Analysis. Masters' thesis. http://libres.uncg.edu/ir/asu/f/Kim,%20Hei-Young_2014_Thesis.pdf

Kim, J. D., Park, J. S., In, B. H., Kim, D., & Namkoong, W. (2008). Evaluation of pilot-scale in-vessel composting for food waste treatment.*Journal of hazardous materials*, *154*(1), 272-277.

Kim, S. H., & Shin, H. S. (2008). Effects of base-pretreatment on continuous enriched culture for hydrogen production from food waste. *International Journal of Hydrogen Energy*, *33*(19), 5266-5274.

Kiran, E. U., Trzcinski, A. P., Ng, W. J., & Liu, Y. (2014). Bioconversion of food waste to energy: A review. Fuel, 134, 389-399.

Kwon, S. H., & Lee, D. H. (2004). Evaluation of Korean food waste composting with fed-batch operations I: using water extractable total organic carbon contents (TOCw). *Process Biochemistry*, *39*(10), 1183-1194.

Lee, J. P., Lee, J. S., & Park, S. C. (1999). Two-phase methanization of food wastes in pilot scale.Applied biochemistry and biotechnology, 79(1-3), 585-593.

Levis, J. W., & Barlaz, M. A. (2011). What is the most environmentally beneficial way to treat commercial food waste? Environmental science & technology, 45(17), 7438-7444.

Levis, J. W., Barlaz, M. A., Themelis, N. J., & Ulloa, P. (2010). Assessment of the state of food waste treatment in the United States and Canada. Waste Management, 30(8), 1486-1494.

Li, M., Zhao, Y., Guo, Q., Qian, X., & Niu, D. (2008). Bio-hydrogen production from food waste and sewage sludge in the presence of aged refuse excavated from refuse landfill. *Renewable Energy*, *33*(12), 2573-2579.

Lin, C. S. K., Pfaltzgraff, L. A., Herrero-Davila, L., Mubofu, E. B., Abderrahim, S., Clark, J. H., ... & Thankappan, S. (2013). Food waste as a valuable resource for the production of chemicals, materials and fuels. Current situation and global perspective.Energy & Environmental Science, 6(2), 426-464.

Lundgren, A., & Hjertberg, T. (2010). Ethylene from renewable resources.Surfactants from Renewable Resources, 111.

Luo, G., Xie, L., Zou, Z., Wang, W., & Zhou, Q. (2010). Evaluation of pretreatment methods on mixed inoculum for both batch and continuous thermophilic biohydrogen production from cassava stillage. Bioresource technology, 101(3), 959-964.

Ma, H., Wang, Q., Qian, D., Gong, L., & Zhang, W. (2009). The utilization of acid-tolerant bacteria on ethanol production from kitchen garbage. Renewable Energy, 34(6), 1466-1470.

MassDEP (2011). MassDEP Streamlining Organic Waste Rules to Foster Clean Energy. Web. 14 Apr. 2013. http://www.mass.gov/dep/public/publications/0611andi.htm

MassDEP (2012a). Organics Study and Action Plan. May 10, 2012

MassDEP (2012b). "MassDEP Organics Subcommittee Meeting Summary." Joe Rasmussen, Ed.D. Loyola Marymount University. 9 Jul. 2012. http://www.mass.gov/dep/public/committee/oscsm79.pdf

MassDEP (2013a). "Composting & Organics.", http://www.mass.gov/dep/recycle/reduce/composti.htm

MassDEP (2013b) On-Site Systems for Processing Food Waste. www.mass.gov/eea/.../massdep-food-waste-final-report.pdf

McDougall, F. R., White, P. R., Franke, M., & Hindle, P. (2008). Integrated solid waste management: a life cycle inventory. John Wiley & Sons.

McLachlan, K. L., Chong, C., Voroney, R. P., Liu, H. W., & Holbein, B. E. (2002, August). Assessing the potential phytotoxicity of digestates during processing of municipal solid waste by anaerobic digestion: comparison to aerobic composts. InXXVI International Horticultural Congress: Sustainability of Horticultural Systems in the 21st Century 638 (pp. 225-230).

Miller, C. (2004, March 1). Food Waste. Retrieved May 16, 2016, from http://waste360.com/mag/waste_food_waste_2

Mirabella, N., Castellani, V., & Sala, S. (2014). Current options for the valorization of food manufacturing waste: a review. Journal of Cleaner Production, 65, 28-41.

Morita, M., & Sasaki, K. (2012). Factors influencing the degradation of garbage in methanogenic bioreactors and impacts on biogas formation. Applied microbiology and biotechnology, 94(3), 575-582.

Nasir, I. M., Ghazi, T. I. M., & Omar, R. (2012). Production of biogas from solid organic wastes through anaerobic digestion: a review. Applied microbiology and biotechnology, 95(2), 321-329.

Neale (2013) Analysis of Biodigesters and Dehydrators to Manage Organics On-site. BioCycle. Vol. 54, No. 10, p. 20.

Neale (2014) Biodigesters and dehydrators: operational experiences. BioCycle. Vol. 55, No. 1, p. 52.

Papanikolaou, S., Dimou, A., Fakas, S., Diamantopoulou, P., Philippoussis, A., Galiotou‐Panayotou, M., & Aggelis, G. (2011). Biotechnological conversion of waste cooking olive oil into lipid‐rich biomass using Aspergillus and Penicillium strains. Journal of applied microbiology, 110(5), 1138-1150.

Papargyropoulou, E., Lozano, R., Steinberger, J. K., Wright, N., & bin Ujang, Z. (2014). The food waste hierarchy as a framework for the management of food surplus and food waste. Journal of Cleaner Production, 76, 106-115.

Papargyropoulou, E., Lozano, R., Steinberger, J. K., Wright, N., & bin Ujang, Z. (2014). The food waste hierarchy as a framework for the management of food surplus and food waste. Journal of Cleaner Production, 76, 106-115.

Parfitt, J., Barthel, M., & Macnaughton, S. (2010). Food waste within food supply chains: quantification and potential for change to 2050. Philosophical Transactions of the Royal Society B, 465,3065e3081.

Pleissner, D., Lam, W. C., Sun, Z., & Lin, C. S. K. (2013). Food waste as nutrient source in heterotrophic microalgae cultivation. Bioresource technology, 137, 139-146.

Rahmani, M., Hodges, A. W., & Kiker, C. F. (2004). Compost users' attitudes toward compost application in Florida. Compost science & utilization, 12(1), 55-60.

ReFED. (2016). A ROADMAP TO REDUCE U.S. FOOD WASTE BY 20 PERCENT. Retrieved March 31, 2016, from http://www.refed.com/downloads/ReFED_Report_2016.pdf

Rynk, R. (1992). On-farm composting handbook.

Sanders, J., Scott, E., Weusthuis, R., & Mooibroek, H. (2007). Bio‑refinery as the bio‑inspired process to bulk chemicals. Macromolecular bioscience,7(2), 105-117.

Shen, F., Yuan, H., Pang, Y., Chen, S., Zhu, B., Zou, D., ...& Li, X. (2013). Performances of anaerobic co-digestion of fruit & vegetable waste (FVW) and food waste (FW): single-phase vs. two-phase. *Bioresource technology*,*144*, 80-85.

Show, K. Y., Lee, D. J., Tay, J. H., Lin, C. Y., & Chang, J. S. (2012). Biohydrogen production: current perspectives and the way forward.International Journal of Hydrogen Energy, 37(20), 15616-15631.

Spencer (2008) A New Generation Of Commercial Disposers. BioCycle. Vol. 49, No. 7, p. 27

Stuart, T. (2009). Waste. Uncovering the Global Food Scandal. Penguin, London.

Tang, Y. Q., Koike, Y., Liu, K., An, M. Z., Morimura, S., Wu, X. L., & Kida, K. (2008). Ethanol production from kitchen waste using the flocculating yeast Saccharomyces cerevisiae strain KF-7. Biomass and Bioenergy, 32(11), 1037-1045.

Taufiq, T. (2010). Characteristics of fresh municipal solid waste.

Tuck, C. O., Pérez, E., Horváth, I. T., Sheldon, R. A., & Poliakoff, M. (2012). Valorization of biomass: deriving more value from waste. Science,337(6095), 695-699.

U.S. Environmental Protection Agency (USEPA). (2016). Food Recovery Challenge (FRC). https://www.epa.gov/sustainable-management-food/food-recovery-challenge-frc#benefits

U.S. EPA. (2009). Municipal Solid Waste Generation, Recycling, and Disposal in the United States, Detailed Tables and Figures for 2008.

UNEP (2009). The environmental food crisis. ISBN: 978-82-7701-054-0.

US EPA (2008). Municipal Solid Waste in the United States: 2007 Facts and Figures. EPA 530-R-08-010. Office of Solid Waste and Emergency Response, Washington, DC.

US EPA (2010). Basic Information about Food Waste. http://www.epa.gov/wastes/conserve/materials/organics/food/fd-basic.html.

US EPA (2013). http://www3.epa.gov/epawaste/nonhaz/municipal/

USCC (2015). Banning Organics from Landfills. http://compostingcouncil.org/landfill-bans-on-organics/

Viturtia, A. M., Mata-Alvarez, J., Cecchi, F., & Fazzini, G. (1989). Two-phase anaerobic digestion of a mixture of fruit and vegetable wastes.Biological wastes, 29(3), 189-199.

Walker, P., Williams, D., & Waliczek, T. M. (2006). An analysis of the horticulture industry as a potential value-added market for compost. Compost science & utilization, 14(1), 23-31.

Wang, F. S., & Lin, H. T. (2010). Fuzzy optimization of continuous fermentations with cell recycling for ethanol production. Industrial & Engineering Chemistry Research, 49(5), 2306-2311.

Wang, G. J., Wang, Z. S., Zhang, Y. W., & Zhang, Y. Z. (2012). Cloning and expression of amyE gene from Bacillus subtilis in Zymomonas mobilis and direct production of ethanol from soluble starch. Biotechnology and Bioprocess Engineering, 17(4), 780-786.

Wang, K., Yin, J., Shen, D., & Li, N. (2014). Anaerobic digestion of food waste for volatile fatty acids (VFAs) production with different types of inoculum: effect of pH. *Bioresource technology*, *161*, 395-401.

Wang, X., & Zhao, Y. C. (2009). A bench scale study of fermentative hydrogen and methane production from food waste in integrated two-stage process. International Journal of Hydrogen Energy, 34(1), 245-254.

Webber, M. (2011). How to make the food system more energy efficient. Scientific American, 29.

Zhang, Y., Banks, C. J., & Heaven, S. (2012). Co-digestion of source segregated domestic food waste to improve process stability. *Bioresource technology*, *114*, 168-178.

Zhuang, Y., Wu, S. W., Wang, Y. L., Wu, W. X., & Chen, Y. X. (2008). Source separation of household waste: a case study in China. *Waste Management*, *28*(10), 2022-2030.

Biographical Information

Arindam Dhar completed his Bachelor in Civil Engineering from National Institute of Technology (NIT), Durgapur, India in May, 2012. After his graduation, he joined Department of Civil Engineering, Southern University Bangladesh as a lecturer before joining The University of Texas at Arlington in Fall 2014 for his graduate studies. His research interests include landfill and solid waste management, landfill mining and numerical modeling.