Food Waste and Sustainability: Quantifying Food Waste Disposal and Evaluating the Environmental Impacts of Technologies and Policies

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Abstract of the Dissertation

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There has been growing interest in establishing food waste prevention and recovery programs throughout the United States. The drive to target food waste stems from increasing concerns about hunger, resource conservation, the environmental and economic costs of food waste, and a general trend in the waste management industry to transition to more sustainable practices. An interdisciplinary systems approach to sustainable food waste management was taken to explore three integrated food waste issues: 1. the quantity of residential, commercial and institutional food waste disposed in the U.S.; 2. the environmental impact of food waste treatment technologies; and, 3. approaches for sustainable waste system planning and the development of food waste management policies. The dissertation begins with a review of the history, contributing factors, impacts, and current practices regarding food waste in the U.S. and globally. Several questions related to food waste and society were explored, particularly the ways in which culture affects food waste generation and how perspectives on food waste vary globally.

The amount of food waste disposed in the U.S. is not well determined, although this information is valuable for policy making and waste management planning. A systematic review and meta-analysis of waste characterization studies was conducted to quantify disposed food waste in a transparent, repeatable, systematic way, and to determine if specific factors drive increased disposal. This approach has benefits over the less transparent modeling used to date to estimate food waste. Sixty-two waste characterization studies were analyzed, representing over 20,000 samples of sorted waste, weighing more than four million pounds. Food waste was found to make up a considerable proportion of the disposed waste stream. The aggregate mean proportion of food waste in U.S. municipal solid waste from 1995 to 2013 was 0.147, which is lower than that estimated by USEPA for the same period (0.179). The proportion of food waste increased significantly with time, and U.S. region significantly affected the proportion of food waste disposed. There were no significant differences in food waste proportions between rural and urban samples, or between commercial/institutional and residential samples. The aggregate mean disposal rate for food waste from 1995 to 2013 was 0.615 pounds of food waste disposed per person per day.

A life cycle assessment was conducted to evaluate the environmental impacts of food waste management technologies for a New York suburban municipality. Four food waste treatment scenarios were modeled, including waste to energy incineration, two types of composting, and anaerobic digestion, to quantify impacts on climate change, eutrophication, acidification, resource depletion, and stratospheric ozone depletion. Results indicated that choices in waste treatment technologies offer opportunities to reduce environmental impacts and produce beneficial end products, particularly through energy and materials recovery. Food waste treatment with anaerobic digestion offered the most impact reductions relative to the other modeled technologies, followed by waste to energy incineration and tunnel composting, and last, enclosed windrow composting. However, other aspects must also be considered when transitioning to more sustainable waste management systems, including economic costs, social priorities, and stakeholder concerns. To address this complexity, a decision and evaluation framework for waste management systems was developed. Emphasis was placed on using the framework to guide system planning to target food waste, particularly food waste prevention. The benefits of food waste prevention policies and the challenges with adopting such programs were examined.

Dedication Page

To my family.

Thank you for your love and support over the years.

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List of Abbreviations

AD: anaerobic digestion

C&D: construction and demolition debris

EASETECH: Environmental Assessment System for Environmental Technologies

EMS: environmental management system

ERS: U.S. Department of Agriculture's Economic Research Service

EU: European Union

FAO: Food and Agriculture Organization of the United Nations

GHG: greenhouse gas

ISO: International Organization for Standardization

LCA: life cycle assessment

MSW: municipal solid waste

PAYT: pay-as-you-throw volume based pricing system for waste

RCRA: Resource Conservation and Recovery Act

USEPA: United States Environmental Protection Agency

U.K.: United Kingdom

U.S.: United States

USDA: United States Department of Agriculture

WRAP: Waste and Resources Action Programme (U.K.)

WTE: waste to energy incineration

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Chapter 1. Introduction

1. Introduction

Food waste has been identified as a significant social, economic, and environmental problem (Nixon 2015, Pearson et al. 2013), and the implications of this element of the solid waste stream have become a topic of growing interest worldwide. Various estimates of how much food is lost or wasted have been made. Gustavsson (2011) estimated that one third of the edible parts of food produced for human consumption is lost or wasted globally. Schneider (2011) estimated that between 10 and 40 percent of the total food produced globally is lost. Lundqvist et al. (2008) found that up to 50 percent of food is lost or wasted. Because of significant data gaps, no consensus has been reached on the actual proportion of food lost and wasted nationally or globally (Parfitt et al. 2010).

Food waste is a considerable component of the world's food system challenges. The global population is quickly growing, urbanizing, and becoming wealthier, which leads to diversification of dietary patterns, and an increase in demand for land, resources, and greenhouse gas intensive foods, such as meat and dairy. These difficulties are exacerbated by the world's changing environmental conditions which cause food production to be unpredictable and increasingly difficult globally (Garnett 2014). It is estimated that continuing population and consumption growth worldwide will lead to an increase in the global demand for food for at least 40 more years, leading to intensified use of natural resources, such as land, water, and energy (Godfray et al. 2010). It is becoming increasingly clear that the many negative environmental effects of food systems must be minimized to ensure enough food is available to feed the world's growing population in an environmentally sustainable way (Tilman et al. 2001). Shifting towards more sustainable food systems is both essential and urgent (van der Werf et al. 2014), and actions are needed throughout the food system on moderating demand, producing more food, improving governance, and reducing waste (Godfray and Garnett 2014). By wasting edible food, all of the resources spent growing, producing, processing, and transporting that food are also wasted, resulting in potentially needless environmental impact (Gustavsson et al. 2011). In addition to reducing the impact of food systems on the environment, reduced food waste and proper waste management can also save economic resources, contribute to food security, and minimize negative impacts of food waste on waste management systems.

1.1 Food Waste as a Component of Municipal Solid Waste

This dissertation focuses on food waste as a component of municipal solid waste (MSW). Food waste, an organic waste, is one of the largest components of MSW in the U.S.; almost none of it is recovered (USEPA 2013). As defined by the United States Environmental Protection Agency (USEPA), MSW includes durable goods, non-durable goods, containers and packaging, food wastes, yard wastes, and miscellaneous inorganic wastes from residential, commercial, and institutional sources (USEPA 2013). Organic waste, including food waste, is different from other wastes because it degrades, enabling the capture of nutrients and energy from it. However, undesirable outputs of degradation may also occur, such as methane emissions to the environment when degradation occurs in uncontrolled anaerobic conditions. As a result, food waste prevention and recovery from the MSW stream is increasingly considered (Lebersorger and Schneider 2011, Platt et al. 2014), and there has been an interest in establishing food waste collection and management programs throughout the U.S. The introduction of appropriate management approaches for organic waste may be critical for promoting waste resource efficiency and opening up new arenas for entrepreneurship and sustainability in waste management (Gomez-Brandon and Podmirseg 2013).

The drive to target food waste stems from growing concerns about hunger, resource conservation, the environmental and economic costs of food waste (Kantor et al. 1997), and a general trend in the waste management industry to transition to more sustainable practices. As society has developed broader social and environmental awareness, people have generally grown concerned about the impact of their dietary lifestyles, including food wastage, on other individuals and the environment (Griffin et al. 2009). Food waste prevention and recovery through alternative management methods have been identified as a means of reducing the environmental impacts of waste generation, and ultimately, leading to more sustainable waste practices. Particularly, energy and nutrient recovery from food waste through biological treatment is gaining in popularity throughout the world (Schott et al. 2013). Food waste recovery is also seen as a way to increase stagnant waste diversion and recycling rates. Generally, diversion and recycling rates have plateaued, possibly because programs for easy to capture materials, such as bottles and newspapers, are well established, and these materials are being recycled. In order to increase rates, another large component of the residual waste stream, such as food waste, must be targeted with new recovery programs. Additionally, food waste prevention is seen as a way to mitigate some of the harmful impacts of food waste in waste management systems (such as by reducing methane emissions in landfills), as well as in global food systems (such as by reducing the impacts of producing food that is not consumed).

Residential, commercial, and institutional food waste diversion programs are being established in the U.S., although they still only contribute to a small reduction in the amount of food waste being disposed in landfills or incinerators. In 2015, BioCycle magazine identified 198 communities in 18 U.S. states offering curbside collection of residential food waste (Yepsen 2015). Programs can also be found at institutional or commercial levels, such as universities and farms. The increase in recovery programs suggests that source separation and management of food waste is a growing trend which could potentially reduce the negative impacts of this part of the waste stream. However, at this time a comprehensive and well-established infrastructure in the U.S. is not present. It is conceivable that not all source separation and recovery efforts of food waste are worthwhile. Options should be thoroughly examined to ensure that actual benefits are achieved from food waste management decisions. Existing data on food waste generation, management, and recovery are limited, making decision making particularly difficult to justify.

A major concern regarding food waste is that the amount of food waste generated and disposed in the U.S. is not well determined. The USEPA materials flow model is used to generate annual estimates of the U.S. waste stream size and composition. Although the USEPA annual reports include estimates of the amount of food waste in the U.S. waste stream, the principles of the model cannot be applied to some organic wastes, such as food and yard waste (Tonjes and Greene 2012). Food waste is not generated by industrial processes where inputs used to create the materials are known and counted, the outputs are tracked, and product lifespans are understood (Nakamura and Kondo 2009). USEPA has acknowledged this, stating that food and yard waste estimates in the model are generated from waste sorts and certain other analyses that are not described (USEPA 2013). Waste characterization sorts, which involve the sorting, classifying, and weighing of wastes, may therefore be the best way to determine the amount of food waste discarded in the U.S. Recent waste sort studies have indicated considerable quantities of food waste being disposed, although these studies have not been comprehensively analyzed prior to this study. Because food waste is increasingly being targeted for prevention and recovery as a soil amendment (Platt et al. 2014), and is one of the organic components of waste that may be converted to energy, a detailed understanding of disposal amounts is valuable. These data can be used to determine if future food waste recovery and prevention efforts significantly change the composition of the residual waste stream.

This dissertation seeks to fill important information gaps by examining several aspects of food waste issues in the U.S., with emphasis on improved policy which can lead to a reduction in harmful impacts of waste and maximize benefits. The results of this research can support more informed decision making by policy makers regarding improved food waste policies, and will guide waste managers on the best means to design and evaluate waste systems.

1.2 Waste Management for Sustainability

Effective waste management is a key component of sustainable development. The Brundtland Commission (1987) defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs." An underlying philosophy of sustainable development is that material and energy usage, as well as pollution, must be restrained so that future generations can thrive (Sikdar 2003). Improved waste management helps achieve sustainable societies because it can directly reduce environmental burdens and natural resource exploitation (Coelho et al. 2012). Awareness of the importance of waste management for sustainability has continued to grow among policy makers, public citizens, and the private sector (Gadenne et al. 2009). As a result, effective waste management and its associated effects are key targets of policies worldwide (Hazel 2009, Jenkins et al. 2009).

Wastes must be responsibly managed for societies to be sustainable (Lehman 2012). Sustainable waste management will become even more important as population growth and rising levels of prosperity in both the developed and developing world lead to increased demand for products and services to be produced, consumed, and eventually discarded (Lehman 2012, Hoornweg et al. 2013). Development and population growth are reflected in the amount of waste generated (Salhofer et al. 2008). As urbanization and waste production increase globally, municipalities face the challenge of how to handle wastes better so as to reduce impacts to human and environmental health. Furthermore, managers and policy makers may also seek to maximize benefits associated with waste management through particular treatment methods such as composting, recycling, and energy recovery (Vergara 2011). Essentially, the fundamental challenge of solid waste management is to minimize potential negative effects while maximizing recovery of useful materials from wastes at a reasonable cost (Bailie et al. 1999).

Environmental benefits from proper waste practices may also indirectly help solve other global environmental issues. Rockstrom et al. (2009) identified nine planetary boundaries which must remain unbreached to maintain a global environment that is conducive for wellbeing. These boundaries represent essential aspects of the complex Earth system which are only able to tolerate specific levels of changes before tipping points are reached with possible catastrophic consequences, and represent the most serious environmental threats humanity faces. They are: climate change; biodiversity loss; excess nitrogen and phosphorus production; stratospheric ozone depletion; ocean acidification; global consumption of freshwater; change in land use for agriculture; air pollution; and chemical pollution. Waste management focused on environmental stewardship can address many of these threats, although indirectly. Food waste prevention, particularly, can address these threats as burdens from agriculture and food production are offset. Bahor et al. (2009) analyzed the effects of improved waste management on climate change worldwide and concluded that global greenhouse gas emissions from a business as usual case (15 percent recycling, 79 percent landfilling, and 7 percent WTE) compared to an improved system (46 percent recycling, 18 percent landfilling, and 36 percent WTE) were considerably higher, indicating that waste management may play a critical role in climate change mitigation. Ultimately, waste systems which strive to achieve high environmental quality can have considerable impacts on the Earth system as a whole.

2. Dissertation Outline

The dissertation is six chapters. This chapter is an introduction providing a broad overview of food waste and sustainable waste management, and a description of dissertation objectives and research significance. The goal of chapter two was to review the history, contributing factors, impacts, and current practices regarding food waste in the U.S. and globally. Chapter three focused on quantifying the amount of food waste disposed in the U.S. in a transparent, repeatable, and systematic way using the extensive dataset of waste characterization sort studies. The powerful statistical and conceptual tools of systematic review and meta-analysis were used as a strong alternative to the obscure methods used to estimate food waste to date. Methods and results were explained in detail, as well as how the methodology can be extended for future analyses. An analysis of how findings compare to USEPA food waste disposal estimates was also performed. In chapter four, the environmental impacts of waste technologies were examined using life cycle assessment (LCA), specifically the Environmental Assessment System for Environmental Technologies (EASETECH) software. The goal was to evaluate the environmental impacts of residual waste disposal for a Long Island, New York (NY) municipality, with focus on how impacts change with the adoption of separate food waste collection and treatment. Four food waste management scenarios were modeled, including waste-to-energy incineration (WTE), two types of composting, and anaerobic digestion (AD), to quantify their environmental impacts on climate change, eutrophication, acidification, resource depletion, and stratospheric ozone depletion. In chapter five, the development of an inter-disciplinary systems analysis tool to facilitate the planning, implementation, and maintenance of sustainable waste management systems was described. It defines key considerations which should be made when designing a waste system and how to monitor system performance over time. The framework was developed using knowledge of waste systems and assessments, current data needs, and an examination of challenges impacting waste systems. In chapter six, policy options for effective food waste management with emphasis on food waste prevention were examined. The framework described in chapter five was applied to food waste management to demonstrate how the framework can effectively help waste system planning, implementation, and design, as well as address key challenges typically faced in waste management.

3. Dissertation Objectives

An interdisciplinary systems approach to sustainable food waste management was taken to explore three integrated aspects related to food waste: 1. the quantity of residential, commercial and institutional food waste disposed in the U.S.; 2. the environmental impacts of food waste recovery technologies; and, 3. approaches for effective sustainable waste system planning and the development of food waste recovery policies. The overall goal of the dissertation was to resolve gaps in the literature concerning food waste. Specific objectives were:

1. Factors Contributing to Food Waste

a) Examine how economic and societal circumstances contribute to food waste

b) Describe behavioral and socio-demographic drivers of food waste generation and disposal

2. Quantification:

    a) Quantify the amount of food waste in the U.S. MSW stream using a transparent, systematic approach

    b) Determine how food waste quantities have changed over time

    c) Identify specific moderators that affect food waste disposal

    d) Assess how food waste estimates from waste characterization studies compare with those estimated by USEPA's materials flow model

3. Environmental Impacts of Food Waste Management:

    a) Identify the environmental impacts of four food waste management technologies

    b) Analyze how LCA can be used to inform policy

4. Food Waste Policy:

    a) Discuss best practices for making socio-environmental decisions for sustainable waste management

    b) Describe ideal food waste management policies. Emphasis is placed on barriers to implementing policies and the means to overcome them

4. Significance of Research

Gustavsson et al. (2011) drew attention to the significant data gaps in the knowledge on global food losses and food waste, and concluded that addressing these issues is an urgent concern. Oelofse and Nahman (2013) also determined that further research on food waste management is essential. Data regarding quantities of food waste generated in the U.S. and globally are limited (Lebersorger and Schneider 2011, Mason et al. 2011), making data-supported policy development difficult. Despite the importance of sound data on food waste generation and disposal rates, there is little ongoing research regarding food waste quantities (Abdulla et al. 2013, Kosseva 2013), and few peer-reviewed or major studies estimating quantities of food waste have been conducted (Buzby and Hyman 2012). In addition to uncertainties regarding the quantities of food waste generated and disposed, knowledge of the environmental impacts of potential management systems is also limited (Bernstad and Jansen

2012a) due to relatively little specific research conducted on food waste recovery and management (Gustavsson et al. 2011). To date, policies promoting greater sustainability in waste management have not spawned equal efforts to develop adequate knowledge about waste generation (Beigl et al. 2008).

This research provided a detailed understanding of the amount of food waste in the U.S. MSW stream which filled critical gaps in the literature. For one, waste characterization sorts were collated and statistically analyzed to determine if there was a consensus on the size of the food waste stream. No one has attempted to do this before and it was unclear whether agreements or differences existed across the multitude of studies. The meta-analytic methodology used for waste characterization sorts was a unique approach which may be extended to quantify other waste stream components. This was the first study focusing on food waste in MSW, making it an expansion beyond more common analyses which look only at general food losses and waste, or waste from specific generator types. Examining the amount of waste that is currently being disposed (to landfills or incinerators) shows the amount of waste that has yet to be recovered from the disposal stream, thus indicating how much waste is available for prevention or alternative treatments. A better understanding of the MSW stream also allows for improvements to key inputs for waste models, such as life cycle assessments, and better, data-driven, policy development and decision making. The estimations of food waste disposal determined here were compared to those estimated by USEPA, which had not been done before. The ability for the USEPA model to accurately define U.S. food waste disposal is unclear, particularly because few details are provided for the estimations, and there is little objective data to support the annual changes in disposal rates for organics (Tonjes and Greene 2012). Here a powerful alternative method was used to estimate disposed food waste which had numerous benefits compared to USEPA's obscure approach.

The environmental impacts of food waste technologies were quantified using LCA for a suburban municipality on Long Island, NY. All residual waste (after recycling) was modeled in four scenarios; one scenario represented current practices (business as usual), with all waste being sent to waste to energy incineration; the alternative scenarios modeled source-separation and alternative treatment of food waste in conjunction with the remaining waste being treated through the business as usual approach. These analyses indicated whether source separation and recovery of food waste provided benefits over business as usual practices. Few LCAs focusing on food waste management have been performed in this context, and no peer-reviewed LCA has been conducted for any of the municipal waste management systems on Long Island to date. This is a non-traditional approach for LCAs, as most focus on food waste alone or whole waste streams, rather than on the residual waste stream. The LCA findings were used to determine the conditions under which food waste recovery is beneficial and how LCA analyses can be leveraged to effectively inform policy. Key questions were answered with regards to making policy decisions from LCA outputs, including how changes in LCA input values affect policy decisions. Most research that has been performed on food waste has been done outside the U.S. (e.g., Andersen et al. 2012, Bernstad and Jansen 2012b), so this work is useful for food waste management in the U.S.

These investigations supported a discussion regarding effective decision making for sustainable waste management, which in turn contributed to a dialogue of idealized food waste management policies. This issue has not previously been thoroughly analyzed in the literature and is particularly timely because food waste is quickly becoming a topic of interest throughout the world, and calls to increase food waste diversion are likely to increase (Levis et al. 2010). Therefore, more research is valuable, especially since there is currently little ongoing research on food waste (Gustavsson et al. 2011). Food waste policies and programs are being implemented in the U.S. and around the world, often absent of quantitative assessments of costs and benefits. Food waste management should be governed by the environmental, social, and economic impacts of specific technologies and policies. Here, these drivers were examined in detail to better understand the future of food waste management. This is placed in a context of improved decision making for sustainable waste management.

Chapter 2. Food Waste Generation, Technologies, and Policies: A Review

1. Introduction

The goal of this chapter is to review the history, contributing factors, impacts, and current practices regarding food waste in the U.S. and globally. First, definitions of food waste are provided, followed by a history of the concept of food waste. Specific socio-demographic, policy, and behavioral factors which lead to food wastage are discussed. The impacts of food system modernization on food waste generation are also explored, particularly impacts related to food system industrialization, urbanization, globalization, and economic growth. Next, the current state of knowledge with regards to food waste quantification is discussed in detail. Last, a review of food recovery technologies and policies are provided, followed by a discussion on the current state of food waste recovery nationally and globally.

2. Food Waste Definitions

Definitions of food waste are not universally agreed upon (Lebersorger and Schneider 2011), which makes studying and quantifying food waste difficult (Buzby and Hyman 2012, Garrone et al. 2014). Different categorizations are made based on what materials are included and excluded, modes of production, and management endpoints (Gjerris and Gaiani 2013). The definition issue is twofold. First, multiple terms have been used interchangeably, such as food loss, food waste, kitchen waste, biowaste, and food and drink waste (Schneider 2013a). Second, often the same terms are used, but with different meanings (Gjerris and Gaiani 2013). This is exacerbated when reports are translated (Schneider 2013a). Defining terms is key to effective and consistent analysis; therefore, I provide an overview of existing definitions (Table 1), with a complete definition of both food loss and food waste as used in this study (Table 2).

Here I focus on food waste rather than food loss because in the developed world, the majority of losses and waste occurs at the consumer and food service levels (NRDC 2012, Parfitt et al. 2010). Because food waste is found in the municipal solid waste stream (MSW), its examination is important for waste system improvements. MSW includes durable goods, non-durable goods, containers and packaging, food wastes, yard wastes, and miscellaneous inorganic wastes from residential, commercial, and institutional sources (USEPA 2013). Therefore, MSW consists of items generated by routine activities of daily life that are used and then thrown away in homes, institutions, and businesses (USEPA 2013, Bailie et al. 1999).

Table 1. Food Waste and Loss Definitions

Author	Year	Definition
Kling	1943	Food waste is the destruction or deterioration of food, or the use of crops, livestock and livestock products in ways which return relatively little human food value.
Food and Agriculture Organization (FAO)	1981	Food waste is all food products allocated for human consumption that are instead discarded, lost, degraded, or consumed by pests at any stage of the food chain.
FAO	2011	Food loss is the decrease in edible food mass throughout the part of the supply chain that specifically leads to edible food for human consumption. Food waste is food losses occurring at the end of the food chain (retail and consumption).
Agricultural and Rural Commission of the European Parliament	2012	Food waste is discarded products of the food supply chain which, for economic or esthetic reasons, or for closeness to the expiry date, despite still being edible and therefore potentially intended for human consumption, in the absence of a possible alternative use, are disposed of.
United States Environmental Protection Agency (USEPA)	2013	Food waste is uneaten food and food preparation wastes from residences, commercial, and institutional establishments. So, food wastes from homes, grocery stores, restaurants, bars, factory lunchrooms, and company cafeterias are included. Pre-consumer food waste generated during the manufacturing and packaging of food are excluded.
United States Department of Agriculture (USDA)	2014	Food waste is a subset of food loss and occurs when an edible item goes unconsumed. Only food that is still edible at the time of disposal is considered waste (Buzby et al. 2014).

Table 2. Food Waste and Loss Definitions Used in this Study

Term	Definition	Drivers	Sectors Included	Examples
Food Loss	Decrease in edible food mass throughout the part of the supply chain that specifically leads to edible food for human consumption	-Infrastructure limitations -Climate and environmental factors -Quality, aesthetic, or safety standards	Production, post-harvest, and processing	-Edible crops left in the field -Food that spoils due to poor transportation infrastructure from factory to supermarket -Food that is contaminated during food processing
Food Waste	Food which was originally produced for human consumption but then was discarded or was not consumed by humans. Includes food that spoiled prior to disposal and food that was still edible when thrown away	-Decisions made by consumers and businesses -Quality, aesthetic, or safety standards	Retail and consumer	-Plate waste -Food that spoils due to poor storage in home or restaurant -Restaurant food prepared but discarded due to lack of demand

I do not distinguish here between avoidable and unavoidable food waste. WRAP (2013) defines avoidable food waste as food that is discarded because it is spoiled or no longer wanted. Most avoidable food waste was, at some point prior to disposal, considered edible. Alternatively, unavoidable food waste is waste that generally is not considered edible in normal situations, such as apple cores and meat bones. The differentiation between avoidable and unavoidable food waste is subjective as some people may consider certain foods edible, while others do not (WRAP 2013). This distinction is important nonetheless as it indicates the degree to which food waste prevention is possible, and it allows study results to be used towards both social and environmental goals (Garrone et al. 2014). However, it is beyond the scope of this study, as here interest was placed on all food waste as a component of MSW.

3. Food Waste History

A history of food waste issues in the U.S. is given in Table 3. The notion of managing food waste is not a new one, but the understanding of it has evolved considerably over time. Today, there is increased attention to the significant issue of food waste, and recent reports have called for waste reduction, with Lundqvist et al. (2008) wanting post-harvest food waste reductions of 50 percent by 2025. Better management of food waste through waste treatment technologies has also been identified as a priority (Lamb and Fountain 2010).

Table 3. Timeline of U.S. Food Waste History

Period	Food Waste Activity
Pre-Industrial (1750-1850)	-Food waste accounted for the majority of household solid waste -These wastes were often fed to animals, usually pigs, because pigs are effective at turning food and plant wastes back into food (Ackerman 1997)
1895	-Atwater (1895) conducted a visual survey of residential New York waste bins and noted upper class areas showed a large portion of food purchased but thrown away; food waste was less in more moderate neighborhoods
1902	-Atwater (1902) found student clubs wasted 10-14% of nutritive value of food; institutions wasted up to 25%
Early 1900's	-Organized waste collection became common in U.S.
World War I (1917-1918)	-U.S. government encouraged pig feeding with food waste as a patriotic means to increase food production
World War II (1941-1945)	-Wartime food scarcities increased attention on food waste (Kling 1943a) -Rationing, which was instituted when voluntary conservation methods proved inadequate, helped control food panics and discouraged wasting food -The U.S. government helped people cope with limited supplies of certain foods (USDA 1943) and encouraged consumers and handlers of food to save every salvageable bit (Kling 1943b) -Williamson and Williamson (1942) noted that considerable food losses and waste was taking place; a large portion of food was wasted by the consumer during food preparation and as plate waste -U.S. Food Distribution Administration (1943) estimated that overall U.S. food wastage was 20-30% of all food production

	-Kling (1943b) estimated that 24% of produced food was lost or wasted -In 1945, FAO was established and listed food loss reductions as a priority
Post-World War II	-U.S. consumer culture evolved from one of thrift (widespread during wartime), to one of abundance and waste because it was no longer patriotic to conserve food and food became less expensive (Bloom 2010)
1950s	-Because pigs fed garbage were particularly susceptible to diseases and food systems were becoming industrialized, regulations prohibited use of raw garbage as animal feed (Ackerman 1997) -USDA began to formally study food waste, generating small, non-representative samples (Adelson et al. 1961, Adelson et al. 1963); they determined household food waste was 7-10% of total calories
1973-1974	-Extensive surveys of household food waste were conducted by the University of Arizona Garbage Project (Rathje and Murphy 2001); they determined food was 9.7% of total household waste output (by weight) in 1973; in 1974, it was 8.9% (Harrison et al. 1975)
1974	-The first World Food Conference (Rome) identified reduction of post-harvest food losses as an element of the solution to global hunger; post-harvest losses were estimated to be 15% and a decision was made to reduce this by 50% by 1985 through the Special Action Programme for the Prevention of Food Losses (in 2010, Parfitt et al. noted no progress had been made towards this goal)
1977	-U.S. General Accounting Office issued a report to Congress titled 'Food Waste: An Opportunity to Improve Resource Use' urging the U.S. to examine food loss and waste
1980-1981	-Food waste was the focal point of Garbage Project research; participant surveys and food waste diaries were integrated into research; households were found to waste considerable amounts of food, but survey participants greatly underestimated the amount of waste (Rathje and Murphy 2001)
1992	-Garbage Project researchers concluded food was a significant portion of household waste (10-15% of all food bought)
1997	-Kantor et al. (1997) published quantitative estimates of food waste across U.S. food system and concluded 25% of food produced in the U.S. (96 billion pounds) was wasted annually
2010s	-Renewed interest in food waste; calls for waste reduction (Lundqvist et al. 2008) and better management (Lamb and Fountain 2010)

4. The Importance of Studying Food Waste

Awareness of food waste and its impact on the economy and environment is growing nationally and globally. This increased interest is partly driven by a better understanding of environmental impacts from food production and waste management, rising food costs, and global food insecurity (Table 4). It is clear that reducing food waste and managing it better can help achieve the priorities of sustainable food systems in terms of social, environmental, and economic factors (Table 5).

Table 4. Motivations for Studying Food Waste

Motivation	Example(s)
Environmental Burdens of Food Supply System	Seepage of fertilizers into environment; excess water consumption for agriculture; GHG emissions from food transport
Economic Losses	Money paid for food that was disposed and not consumed
Food Insecurity	People lack sufficient food or access to reliable food sources
Environmental Burdens of Food Waste Disposal	GHG and toxic emissions from disposal technologies

Table 5. Key Priorities of Sustainable Food Systems

Social	Environmental	Economic
-Availability and accessibility to sufficient amounts of food and good nutrition	-Minimization of resource (i.e. land, energy, water) use for food production and transportation -Minimization of environmental impacts of food production and food waste disposal	-Efficient use of economic resources for food production -Optimization of food purchases

4.1 Environmental Impacts of Food Production, Storage, and Transportation

There is growing recognition that there are significant environmental burdens associated with the food supply system, which includes food production, packaging, and distribution (Jones 2002). The current system is not environmentally, socially, or economically sustainable. Producing food affects the environment to the detriment of humans, animals, plants, and ecosystems generally (Gjerris and Gaiani 2013). There has been a decadal shift in demand from local and seasonal foods toward imported, non-seasonal fruits and vegetables, increasing transportation and energy use. More food processing also has led to increased energy and material inputs (Kosseva 2013). The increased demand for resource intensive foods, such as meats, makes the environmental impact greater.

Food production and distribution requires large amounts of energy and other resources (Cuellar and Webber 2010). Key environmental risk areas include water, soil, and air. Food production can contribute to water pollution and eutrophication, particularly due to the seepage of nutrients, such as manure and fertilizers, into the broader environment. Agriculture is the largest human use of water so it is a great consumer of a limited resource (Lundqvist et al. 2008). Agriculture may lead to sediment transport and deposition downstream, as well degradation of aquifers (Trautmann et al. 2015). Food supply chains can also have negative emissions to air, including pollution caused by agricultural machines and food transport vehicles. The production and transportation of edible food has been shown to contribute to greenhouse gas emissions (Weber and Matthews 2008). Soils can be harmed from food supply chains. Direct effects of food supply systems on the land include soil erosion, nutrient depletion (Nellemann et al. 2009), on and off site pollution (Trautmann et al. 2015), deforestation, desertification, and biodiversity loss. A large percentage of the world's land area is in agriculture; approximately 51 percent of U.S. land is used for growing food (USDA 2014b). Land use changes resulting from agriculture can result in biodiversity loss, loss of natural ecosystems, and overall ecological degradation (Pretty et al. 2005).

By wasting edible food, all of the resources that went into growing, producing, processing, and transporting that food are also wasted, resulting in potentially needless environmental impact (Gustavsson et al. 2011). At a global level, Kummu et al. (2012) estimated that one quarter of the freshwater and one fifth of cropland and fertilizers are used to produce food that is lost or wasted. In the U.S., the production of wasted food requires the expenditure of over 25 percent of the total freshwater used in the U.S., about 300 million barrels of oil (Hall et al. 2009), and represents two percent of annual energy consumption (Cuellar and Webber 2010). Venkat (2011) estimed that 112.92 million metric tons carbon dioxide equivalent (MMT CO_2 eq.) per year were emitted from the production, processing, and disposal of avoidable food waste in the U.S.

The impact of food waste on the environment is of particular concerning because population growth and consumption patterns will continue increasing worldwide, leading to higher global demand for food for at least 40 more years, amplifying the pressures of food systems on the environment. Thus, it is critical that the impact of food systems on the environment be reduced, yet still produce enough food to feed the world (Tilman et al. 2001, Godfray et al. 2010). One means of reducing the environmental impact of food systems on the environment is to minimize the amount of food that is produced but is discarded (Godfray et al. 2010).

4.2 Economic Losses

Table 6 provides recent estimates of the financial costs of wasted food.

Table 6. Economic Costs of Food Waste

Country	Year	Estimate [a]	Sectors Included	Reference
Global	2013	$750 billion/year	All sectors (seafood excluded)	FAO 2013a
U.K.	2012	$18.3 billion/year, $689/household/year	Household	WRAP 2013
U.S.	2011	$197.7 billion/year, $643.3/person/year	Avoidable distribution, retail & consumer waste	Venkat 2011
U.S.	2010	$161.6 billion/year, 1,249 calories/person/day	Avoidable retail & consumer	Buzby et al. 2014
Canada	2010	$21.1 billion/year	All sectors	Gooch et al. 2010
U.S.	2008	$165.6 billion/year, $390/person/year	Avoidable retail & consumer	Buzby and Hyman 2012

[a] Estimates given in currencies other than U.S. dollars were converted to U.S. dollars (exchange rate as of March 2015)

The economic impact of throwing food away affects all the individuals and organizations involved in the food supply chain.

4.3 Food Insecurity

Food security, the availability of and access to sufficient and healthy foods and good nutrition, is imperative for the wellbeing of individuals and nations (Soussana 2014). Although there appears to be sufficient food available to feed the world's population, one billion people are food insecure (Kosseva 2013). From 2011 to 2013, 842 million people (12 percent of the global population) were unable to meet their dietary energy requirements (FAO 2013b). In the U.S., 14.5 percent of households were food insecure at least some time in 2012, meaning they lacked access to enough food for an active, healthy life for all household members (Coleman-Jensen et al. 2013). Due to this high prevalence of food insecurity, food wastage has an important ethical dimension (Gjerris and Gaiani 2013). If food resources were managed better and wastes were minimized, resources could be used to help feed the hungry, such as by diverting food through charitable donations. Furthermore, food loss and waste amplify the environmental impact of food production along the entire supply chain by requiring more production than is needed based on market demand. Therefore, reducing food waste, while maintaining current production levels, could help global food needs today, and in the future as population rises (Stuckey et al. 2013). If less food were wasted, fewer resources would be required to produce food that is not consumed, and these agricultural lands and resources could be liberated for other uses, such as growing food for those that are food insecure.

Reducing food waste will improve future food availability in the context of global population growth and increasing resource scarcity (Pearson et al. 2013, Godfray et al. 2010, Buzby et al. 2014). The United Nations estimate that the world population will reach 9.3 billion by 2050 (United Nations 2013) and this growth will require an increase in food production by about 70 percent, including an additional annual consumption of nearly one billion metric tons of cereals for food and feed, and 200 million metric tons of meat (FAO 2009). To produce enough food to sustain this high population, pressure will be increased on agricultural land and other limited resources. It is necessary to develop ways to provide more food with fewer inputs so that the world's food system can deliver better nutritional outcomes at a smaller environmental cost (Garnett 2014). Sustainability throughout the entire food system should be increased by moderating demand, producing more food, improving governance, and reducing waste (Godfray and Garnett 2014).

4.4 Environmental Impacts of Food Waste Disposal

Food waste may have negative environmental impacts at the end of its life depending on how it is managed. In landfills, food waste converts to methane, a greenhouse gas with a global warming potential 25 times greater than carbon dioxide (CO_2) on a 100 year time scale (IPCC 2007). Although one quarter of U.S. landfills capture methane to create energy, fugitive emissions (emissions prior to the installation of gas collection systems) and landfills without collection systems cause landfills to be the third largest source of anthropogenic methane in the U.S. (after natural gas/petroleum systems and enteric fermentation) (USEPA 2011). Because food waste tends to degrade faster than other landfilled organic materials, has a high methane yield, and does not contribute to considerable biogenic sequestration in landfills, its diversion from landfill should be a priority (Table 7). However, food waste can also generate benefits if managed in other ways, or in landfills with efficient gas collection systems. The biodegradability of food waste enables alternative disposal practices, such as composting or anaerobic digestion, which can create useful end products (energy and compost), and the production of energy using landfill gas to energy collection technologies.

Table 7. Material Properties for Selected Organics

Material	Field Decay Rate (yr^{-1})	Methane Yield (m^3 dry Mg^{-1})	Carbon Sequestration Factor (kg C dry Mg^{-1})
Food Waste	0.144	300	80
Newspaper	0.033	74	420
Office Paper	0.029	217	50

Adapted from Levis and Barlaz 2011b

Preventing food waste and diverting it away from disposal (landfill or incineration) is thought to be a means to improve the overall environmental performance of waste management systems. Throughout the U.S., there has been considerable concern over stagnant recycling and diversion rates. Recycling programs spread across the country quickly throughout the late 1980s and 1990s (Ackerman 1997) and recycling rates continued to increase until about the early to mid-2000s when they began to level out. Targeting and diverting food waste may be a means to increase these stationary rates and improve the overall environmental performance of waste management systems.

5. Societal Drivers for the Generation of Retail, Consumer and Institutional Food Waste

There are many drivers for food waste generation at the retail and consumer levels, although detailed information on the exact causes is limited (Lebersorger and Schneider 2011). In the developed world, particularly the U.S., increases in the volume, caloric density, low cost relative to other expenses, and constancy of food supplies have led to a growing number of people discarding more food (Sobal 1991). There tends to be little understanding regarding what food is, where it comes from, and what its production entails (Stuart 2009). There are some constant factors for food waste generation; others, however, have greater variation, such as the socio-demographic characteristics and cultural traditions leading to activities that contribute to food waste (Buzby and Hyman 2012). Culture and personal choice affect decisions regarding what is too good to throw away, and these perceptions can change over time. Striking differences in attitudes towards food and food waste have been documented across nations (Stuart 2009). Therefore, food waste generation is a function of cultural, personal, political, geographic, and economic forces that influence behavior in specific situations (Pearson et al. 2013) and it may differ from person to person, year to year, or from society to society.

5.1 Modernization of Food Systems

Modernization in food supply chains is associated with industrialization, economic growth, urbanization, and globalization. It is manifested through dietary transitions and influences the amount and type of food that is wasted (Table 8). Countries move through nutritional transitions and food supply changes at different rates, often directly related to economic development (Drewnowski 1999, Popkin et al. 1993). Those cultures which place emphasis on food as a finite, valuable resource that is to be cherished, may modernize at slower rates and ultimately have differing wastage patterns (Stuart 2009).

Table 8. Modernization's Effects on Food Systems

Factor	Description	Effects on Food Systems
Industrialization	Transition from food production and preparation at home to large scale operations and factories	- Increases distancing of people from food production and preparation - Increases food preparation outside the home - May reduce food costs - Contributes to abundance and variety of food
Economic Growth	Increase in disposable income	- Increases diet diversification, particularly a transition away from traditional foods - May cause reductions in disposable income spent on food
Urbanization	Population shift from rural to urban areas which requires the extension of food supply systems to feed urban populations	- Increases diet diversification - Increases distancing of people from food production
Globalization	Shift from local to global food sources; transition of dietary patterns away from traditional ways towards global trends	- Increases diet diversification away from local foods - Increases distancing of people from food production

Industrialization of food systems, which results in a transition of food production and preparation from the home to factory and from handcraft to purchasing (Strasser 1999), affects the foods that people consume, the type and quantities of food waste, and contributes to increased physical distancing of people from food production and preparation. In areas with industrialized food systems with large amounts of food processing, people often purchase pre-made foods, or canned and frozen vegetables. As a result, pea pods and corns husks, for example, become industrial wastes, while packaging becomes more common in household waste. In industrialized food systems, consumers often purchase pre-cut meats, such as chicken legs, so there are no other components of the chicken to be disposed as waste at the consumer level; the other parts of the chicken are utilized or disposed by industry during the chicken processing.

Increased frequency of eating at restaurants and consumption of takeout food (commercially prepared but consumed at home) (Sobal 1999) have been observed in the developed world. This is partly due to the dramatic rise of two-earner households, leading to little available time for food selection and preparation in the home. As food preparation and consumption is increasingly accomplished in restaurants, some shifts in food waste from homes to the commercial sector may occur. It is estimated that almost half the U.S. food budget is spent eating away from home; USDA (2013) estimates that in 2012, $672.2 billion was spent for food prepared in the home and $629.7 billion was spent on food outside of the home. This is a dramatic change from the early-twentieth century where almost all food expenditures were spent on food prepared within the home. In 1929, $15.3 billion was spent on food in the home, and $3.5 billion was spent on food from outside (USDA 2013). Adults tend to be less likely to waste food that they prepared themselves or that a loved one prepared. In cultures based on handwork, handmade things are valuable as they embody many hours of labor. People who have not created or prepared something themselves, or watched a loved one do so, value labor less than those who have, and therefore, are more likely to throw it away (Strasser 1999). As food preparation and consumption is increasingly done in restaurants, factories, or supermarkets, there is likely to be shifts in the types and quantities of food waste generated by residences, industry, and commercial establishments.

Higher incomes have generally been associated with the consumption of a more varied diet (Drewnowski 1999). Growth in household incomes is associated with a decline in starchy food staples and a diversification of diet towards more meats, dairy, fish (Parfitt et al. 2010), and poultry (Fischler 1999), per Bennett's Law (food share of starchy staples decreases as income increases) (Bennett 1941). This worldwide trend with increases in consumption of protein and energy rich foods, and convenience foods, and decreases in rice consumption has been documented (Pingali 2004). Particularly, Asian diets are shifting towards more Western foods (Pingali 2004). Western diets have been associated with greater food waste and a greater drain on environmental resources (Lundqvist et al. 2008). Rathje and Murphy (2001) point out that diet diversification may lead to more food waste, and the more repetitive the diet, the less food wasted. They found census tracts with mostly Mexican-American families had less food waste because the ingredients for Mexican food are consistent, making it easy to incorporate leftovers into new meals and staple ingredients are used in almost every meal. In restaurants, larger menus lead to more waste because there are additional ingredients to manage.

As incomes rise, people may be able to waste food because food expenditures are not significant portions of their income. In wealthy countries, such as the U.S., food is relatively inexpensive compared to other expenses (e.g., housing) and people can afford to waste food (Pearson et al. 2013). FAO suggest that the careless attitude of consumers who can afford to waste food is a large contributor to household food wastage (Gustavsson et al. 2011). The proportion of U.S. household income spent on food in the home has steadily declined as people have gotten wealthier, food prices have decreased, and the cost of other necessary items have increased. USDA determined that in 1960, Americans spent 18 percent of their disposable personal income on food; the percentage steady declined and in 2013, it was less than ten percent (USDA 2013a). In poorer countries, however, expenditures on food are still high. For example, in Pakistan 48 percent of disposable income was spent on food in 2012; in Cameroon, it was 46 percent (USDA 2013a).

Urbanization requires extensions of food supply systems (Parfitt et al. 2010) and leads to diet diversification and a disconnection from food sources, which ultimately may increase food waste. Urbanization has increased substantially in the U.S.; in 1790, five percent of Americans lived in urban areas, by 1890 it was 35 percent, and in 2010, it was 81 percent (U.S. Census, 2012). Urbanization is expected to continue increasing globally (United Nations 2013). Concentrated, population dense urban food systems are different from those of dispersed, low density rural systems (Solomons and Gross 1995). There are far fewer farms and farmers in urbanized areas, so fewer people interact directly with agricultural processes or live near places where food is produced, hindering knowledge about food origins. This promotes disconnections from food (Parfitt et al. 2010), so that people have no sense of what their food is made of or how it was produced (Fischler 1999). Since food sources are not local, there are more opportunities to market diverse foods, different from those grown locally. Lebersorger and Schneider (2011) found residual waste from urban Austrian households contained significantly more food waste than rural areas.

Food systems have changed due to the shift from local to regional to global foods in terms of size, scope, variety, and volume. Globalization means the linkage and integration of previously local, national and regional phenomena into organizational arrangements at a global scale (Sobal 1999). Food supply globalization was made possible by social and technological changes occurring after food supply industrialization (Robertson 1990). New dietary patterns reflect global patterns and may differ significantly from traditional food practices (Pingali and Khwaja 2004), particularly because non-local foods are available for consumption and there is an overall increase in the range and quantities of available foods. Globalization has been associated with the consumption of fewer locally produced plant foods and more imported and processed foods, particularly animal products (Sobal 1999). Food now travels long distances (Pretty et al. 2005) and to more supermarkets in place of small, local markets. Changes in diets spurred by globalization affect the types of foods that are disposed; people also may be more likely to waste food as they do not have a deep connection and understanding of it.

5.2 Cultural Factors

Culture plays a fundamental role in shaping food, eating, and nutrition (Sobal 1998), as well as waste generation. The amount of food a society wastes is dependent on habit and attitudes. People from different cultures regard different foods and food parts as edible, and throw different parts away (Strasser 1999). Pollan (2007) points out that some cultures, particularly the U.S. and Australia, have weak food traditions of their own, meaning there are few longstanding rules and rituals about what to eat and when to eat it, and there are weak connections between the production and preparation of food and its consumption. Bloom (2010) has argued that the U.S. has an unhealthy relationship with food, and overall, the U.S. food culture places little value on food, leading to waste.

Other societies have strong appreciation for food, including production and preparation. Countries such as France and Italy have deep food cultures which have been developed over long periods of time. These cultures are resilient to change (or at least change slowly) primarily due to strong values surrounding what foods can be grown during certain seasons, and how foods are prepared. Food ingredient transformations into material for human consumption are central to deep food cultures. Many cuisines depend on the longevity of traditional recipes and cooking techniques (Conveney et al. 2012). Deep food cultures may be less affected by changes brought on by modernization of the food supply system and may show different wastage patterns.

5.3 Socio-Demographic Factors

Surveys of attitudes and behaviors have shown some correlations between food wasting behaviors and certain socio-demographic characteristics (Pearson et al. 2013), although there is no clear consensus regarding which socio-demographic factors relate to more waste. Age has been shown to affect food waste generation, with young people wasting more than older people (Hamilton et al. 2005, Cox and Downing 2007, Quested and Johnson 2009). In Australia, food waste fell sharply as age increased; among 18 to 24 year olds, 38 percent of respondents wasted more than $30 (Australian) on fresh food per day, compared to seven percent of people aged 70 and up (Hamilton et al. 2005). In the U.K., people over age 65 wasted considerably less food than the rest of the population (approximately 25 percent less when household size was controlled for). These older participants felt that wasting food was wrong, which may be based on the fact that many people of this age group experienced austerity and food rationing during World War II, establishing attitudes against wastefulness (Quested et al. 2013). It is unknown if current young people will waste less as their knowledge, attitudes, and lifestyle change as they age (Pearson et al. 2013).

Family composition and household size significantly affect food waste generation (Wenlock and Buss 1977, Vangarde and Woodburn 1987). Households with children waste more than households without children (Hamilton et al. 2005, Cox and Downing 2007). One common cause for food waste in Swedish households was that children often did not want to finish their food. Larger households waste less per capita than smaller households (Williams et al. 2012), especially those where people live alone. Koivupuro et al. (2012) found no significant difference in waste per capita based on household size, but people that lived alone generated the most waste per capita. In particular, women that lived alone generated the most food waste per capita.

Lower food loss has been found in low-income compared to high-income households (Cox and Downing 2007), but more studies found little or no correlation between income and food wastage (Wenlock et al. 1980, Van Garde and Woodburn 1987, Koivupuro et al. 2012).

5.4 Personal Behavioral Factors

Food wastage is not the result of a single behavior, but combinations of multiple behaviors (Quested et al. 2013). Cultural, political, economic, geographic, and socio-demographic factors may cause the behaviors, but so can personal preference and values. At the retail and institutional levels, food is generally wasted due to choices regarding quantities of available food and visual qualities of food. Specific causes include (1) un-purchased specialty holiday food; (2) damaged packaging; (3) damaged or inadequately prepared items; (4) overstocking or over-preparation of food; (5) routine kitchen preparation waste; and (6) out-grading (Buzby and Hyman 2012). Appearance quality standards cause retailers, particularly supermarkets, to out grade foods due to rigorous quality standards concerning weight, shape, and appearance (Gustavsson et al. 2011). Many grocers take pride in beautiful food displays with uniform, flawless food, which require the culling of even slightly imperfect items. Overstocking also is an issue, particularly in supermarkets, because grocers would rather put more stock out than run out of items, as they believe customers want to see full shelves (Stuart 2009).

In food service, plate waste is a significant contributor to food waste (NRDC 2012), and results from large portion sizes and undesired accompaniments. Portion sizes are increasing inside and outside the home (Wansink and van Ittersum 2007, Wansink and Payne 2009, Wansink and Wansink 2010). Portion sizes began to rise in the 1970s, and then increased sharply in the 1980s and continued to climb in the 1990s. Portion increases have been seen in supermarkets, where the number of items in larger sizes has increased 10-fold between 1970 and 2000 (Young 2003). The average sizes of certain foods, such as bagels and muffins, have increased significantly over the past 20 years. These large portions encourage both waste and obesity (Young and Nestle 2002). Kallbekken and Saelen (2013) found that reducing the physical size of plates in hotels reduced food waste by 19.5 percent.

Consumer behavioral choices cause food wastage at the household level through the interaction of aspects of food's journey into and through the home: planning, shopping, storage, preparation, and consumption (Quested et al. 2013). Poor planning at the shopping stage leads to over-provisioning and impulse or bulk purchases (Evans 2012, Koivupuro et al. 2012), which are significant contributors to food waste (Pearson et al. 2013). In high income countries, food is commonly purchased without much thought as to how it will be used (Gustavsson et al. 2011). In developing countries, consumers generally buy smaller amounts of food each time they shop, often just enough for meals that day (Pearson et al. 2013).

In the home, wastes may be generated due to preparing too much food (Koivupuro et al. 2012), or preparing food inadequately. In the U.K., 40 percent of household food waste was due to the preparation and serving of more food than could be consumed (Quested and Johnson 2009). Over-provisioning is both intentional and unintentional, as cooks may find it difficult to estimate how much to cook, but they also would rather prepare too much food than not enough (Pearson et al. 2013). Portion sizes in the home, as measured in the sizes of bowls, glasses, and dinner plates, and serving sizes as presented in cookbooks, have been increasing. The serving size of some entrees increased by as much as 42 percent in the 2006 *Joy of Cooking* cookbook from recipes in the first (1931) edition (Wansink and Payne 2009). People may lack the skills to prepare food well, or to reuse leftovers. Food spoilage due to improper or suboptimal storage, poor visibility in refrigerators, and partially used ingredients, leads to wastage (NRDC 2012). A survey of U.K. households found 47 percent more fresh food was wasted compared to frozen foods because fresh food spoils faster (Martindale 2014). Another U.K. study found that more than half of food waste occurs because food was not used in time (Quested and Johnson 2009), possibly due to confusion over "use by", "sell by", "enjoy by", and "best by" date labeling (Van Garde and Woodburn 1987, Quested and Johnson 2009). In the U.S., there are no federal standards on the presentation and meaning of date labels on food. State rules vary in coverage and what the dates mean, which leads to consumer confusion (Kosa et al. 2007), and often results in safe, edible food being thrown away. This confusion and general misconceptions about food safety and high sensitivities to food safety are contributors to food waste (Pearson et al. 2013). Studies also find that consumers are unaware or unconcerned about food waste (Buzby et al. 2011, Pearson et al. 2013). Behavioral factors contributing to food waste at the retail and consumer levels are summarized in Table 9.

Table 9. Behaviors Contributing to Food Waste at Retail and Consumer Levels

Factor	Description
Over Preparation / Large Portion Sizes / Undesired Food	Excess food that is prepared but that is not consumed (includes plate waste)
Inadequate Food Preparation / Lack of Food Preparation Skill	Food that is prepared incorrectly (e.g., burnt) or poorly (e.g., poor tasting food); inability to reuse excess food or incorporate left-overs into a new meal
Defects in Food or Food Packaging	Imperfect qualities of the food (such as bruising); damaged food packaging (e.g., dented cans) (includes out-grading)
Over Stocking	Excess food that is purchased but not consumed/sold (either at retail or consumer levels)
Spoilage/Food Not Used in Time / Confusion Over Date Labels	Food that spoils before it can be consumed/sold; food that is believed to be inadequate for consumption based on personal preferences, date labels, or conceptions about food safety
Routine Kitchen Preparation Wastes	Non-edible food components that are disposed of as part of routine kitchen preparation (e.g., apple cores)

5.5 Policies Driving Food Waste Generation

There are policies which contribute to retail and consumer food waste by mandating food disposal under certain conditions or by preventing its redistribution elsewhere. These policies aim to achieve some overall benefit (food safety or enhanced nutrition), but they may also lead to increased food wastage. Furthermore, litigation concerns may discourage the reuse or redistribution of edible food. As a result, there is tension between the need for food safety and nutrition and the desire to reduce food waste (Watson and Meah 2012).

Examples include the USDA's 2012 Nutrition Standards in the National School Lunch and Breakfast Program, which emphasized nutritional quality improvements for student meals. This policy has been criticized for leading to substantially more food waste because students do not like the new meals and are throwing away fruits and vegetables that they are required to take (Jalonick 2014). At one elementary school after the implementation of the policy, 45 percent of served food and beverages were discarded by students (Byker et al. 2014). Cohen et al. (2014) evaluated plate waste at several schools before and after the 2012 standards were implemented, and found substantial amounts of food waste both before and after the 2012 policy, so it is unclear whether the standards were the cause of food wastage. Because whole grain foods are another contentious issue, a bill was proposed in 2014 to modify the meal standards, particularly regarding the amount of whole grains required in meals (Jalonick 2014).

The U.S. Food and Drug Administration sets federal calls for food safety, which are promulgated at the state and local levels as well. Food safety inspections or food labeling requirements mandate the disposal of food that is not allowed to be sold or consumed, such as food that is improperly labeled or inadequately stored. Approximately 48 million pounds (three percent) of imported fresh produce inspected under the Agricultural Agreement Marketing Act are rejected or destroyed annually because they do not meet standards (USDA 2014a). USDA and the European Union (EU) have recognized that food safety policies contribute to waste, but consider human health protection the primary concern. Still, both governmental organizations have vowed to reduce food waste (European Commission 2014, USDA 2014a). The USDA is working to streamline donation procedures for wholesome misbranded or non-standard food that is fit for human consumption to redistribution agencies, and has spearheaded several food waste reduction initiatives, such as through tax incentives for donors and liability protection. These efforts include the Bill Emerson Good Samaritan Food Donation Act, U.S. Federal Food Donation Act of 2008, and Internal Revenue Code 170(e)(3).

6. Quantifying Food Waste

Definitional issues, the absence of sound quantification methods, and a general lack of imperative reasons or political will have led to considerable data gaps with regards to food waste generation and disposal quantities. A range of diverse methodologies have been used to quantify food waste, all of which have some drawbacks. The specific quantification methods used depend on the purpose of analysis. Some approaches, such as waste characterization sorts and materials flow modeling, attempt to quantify the amount of food waste disposed in MSW. Other methods focus on overall generated food waste amounts from specific sectors, such as households or restaurants, or aim to link disposal amounts with behavioral actions. These methods include food diaries, qualitative surveys/interviews, and food supply and nutrition data analyses. Table 10 presents some recent published countrywide and global estimates of food loss and waste and illustrates the diversity in scope, scale, and methodologies used for quantification.

Table 10. Recent Estimates of Food Loss and Waste

Reference	**Estimate** [a]	**Location**	**Method**	**Food Loss** [b]	**Food Waste** [b]
Pekcan et al. 2006	816.4 grams/household/day	Turkey	FAO food supply data, household expenditures, & survey		√[c]
Lundqvist et al. 2008	Up to 50% of total production	Global	Food supply and loss data from Smil 2000	√	√
WRAP 2009	8.3 million tonnes/year (22% of purchases)	U.K.	Food diary, composition analysis, and local data		√[c]
Hall et al. 2009	40% of total food supply (1,400 calories/person/day)	U.S.	FAO food supply data & human energy expenditure model	√	√
DEFRA 2010	15% of edible food & drink purchases (16% of edible calories)	England	Food purchasing data and WRAP 2009 waste estimates		√[c]
Buzby et al. 2011	29% of available food supply	U.S.	USDA food supply data & loss factors		√[d]
Gustavsson et al. 2011	33% of total food production (1.3 billion tons/year)	Global	FAO food supply data & loss factors developed by the authors	√	√
Koivupuro et al. 2012	23 kilograms/person/year	Finland	Food diary		√
Kummu et al. 2012	25% of total food production (614 kcal/person/day)	Global	FAO food supply data & loss factors from Gustavsson et al. 2011	√	√
WRAP 2013	4.2 million tonnes/year	U.K.	Food diary, composition analysis, and local data		√[c]
Beretta et al. 2013	48% of total calories	Switzerland	Mass & energy flow model	√	√
USEPA 2014f	34.69 million tons/year	U.S.	Materials flow model		√[e]
Oelofse & Nahman 2013	9.04 million tonnes/year (177 kg/person/year)	South Africa	FAO food supply data & loss factors from Gustavsson et al. 2011	√	√
Buzby et al. 2014	31% of available food supply (133 billion pounds)	U.S.	USDA food supply data & loss factors		√[d]

[a] Estimates as reported in each study. Exact definitions of food loss and waste used may differ from the definitions used here. Some of these differences are noted.
[b] Food loss and waste are defined in Table 2
[c] Only residential waste included
[d] Only retail & consumer waste included
[e] Only food waste disposed in the MSW stream included

Recent studies have sought to quantify food wastage, although data are still fragmentary (Griffin et al. 2009, Parfitt et al. 2010). Schneider (2011) estimated that between 10 and 40 percent of the total global food production is lost for human consumption for different reasons. Lundqvist et al. (2008) estimated that as much as half of all food grown is lost or wasted before or after reaching consumers. An FAO study estimated that one-third (about 1.3 billion tons per year) of all food produced for human consumption is lost or wasted (Gustavsson et al. 2011). Global (Parfitt et al. 2010), national (WRAP 2011, Buzby et al. 2014), and regional (Griffin et al. 2009) studies exist. Still, Lebersorger and Schneider (2011) claim there are few authoritative, systematic, or comparative data on food waste quantities. Parfitt et al. (2010) note that the validity of past estimates are questionable as they tend to use the same limited and outdated datasets, and due to the significant data gaps, no consensus has been reached on the actual proportion that is lost or wasted nationally or globally.

6.1 Waste Characterization Sorts

Waste characterization sorts are the most common quantitative method to analyze waste streams (Koivupuro et al. 2012). They involve the representative sampling, sorting, and weighing of wastes originating in a target waste shed (Bobman and Culbertson 2010) to determine the proportion of various materials in the samples. Data from waste sorts can influence decisions for recovery efforts, such as recycling and composting, and they may be used to assist in planning, policy development, and infrastructure sizing. Inherently, understanding waste composition is important for solid waste management (Bobman and Culbertson 2010). Sorts may be performed at waste generation points or at management sites, although sampling at management sites is generally performed for municipal-scale studies (Bailie et al. 1999). Studies at management sites tends to be less expensive and more valid than that from generation points (Bailie et al. 1999), and they tend to follow a standard methodology, unlike studies done at generation points which have no international standard methodology and lack consistency between studies (Dahlen and Lagerkvist 2008). Analyses done at generation points tend to be site and industry specific (limiting their generality) and often are only available internally (not formally published or otherwise available).

Most studies performed at management sites are full-scale characterization studies of residual waste which involve sorting samples of disposed waste (the fraction remaining after the removal of materials for recycling and composting) into well-defined categories. These studies involve representative sampling of wastes, as it is neither practically feasible nor desirable to perform waste sorts on all disposed waste. Procedures for waste composition analysis include a widely cited protocol developed by American Society for Testing and Materials (ASTM D 5231-08) (ASTM 2008, Bobman and Culbertson 2010). It outlines particular details regarding: (1) determining the number of samples needed to achieve reasonably low levels of errors for the mean composition estimates; (2) selecting representative distribution of incoming trucks containing wastes from the targeted waste shed; (3) obtaining a representative sample of waste from tipped loads; (4) sorting the samples into individual material categories and weighing the relative contribution of each constituent to the overall samples; and (5) calculating the mean, standard deviation, and confidence intervals for the sample data. The general principles of the ASTM method are usually followed for waste characterization sorts (Staley and Barlaz 2009); variants usually are very similar to the standard (Bobman and Culbertson 2010). Waste sorts may be the best means to accurately quantify waste streams (Kelleher and Robins 2013, Evans 2012, Stuart 2009) because they involve standard methods and reduce subjectivity found in survey studies, and avoid assumptions needed for modeling approaches.

Although waste sorts enable an objective measurement of the amount of food waste in a waste stream, they do not provide detailed pictures of specific types of food waste or reasons behind the disposal of the food (Koivupuro et al. 2012). Sampling may also lead to uncertainties such as skewing due to atypical circumstances or specific local situations. Waste sorts at management points only indicate the amount of food discarded as MSW and that was collected through traditional waste management systems; they exclude waste disposed via other disposal routes, such as food disposals, home composting, charitable donations, or food fed to animals. They also are labor intensive.

6.2 Materials Flow Modeling

Franklin Associates, Ltd. regularly estimates the quantity of MSW generated and disposed in the U.S. under contract for the USEPA using the materials flow methodology (USEPA 2013). This methodology relies on industrial production data for materials and products in waste streams. Waste generation is determined by making specific adjustments to the production data, such as for imports/exports and product life spans (USEPA 2013). Essentially, the model is based on the assumption that everything is eventually discarded or otherwise managed (Bailie et al. 1999). The model breaks the overall estimate into specific waste categories, including food waste, and by how much generated waste is treated by specific approaches, such as recycling or composting. This is the only large scale study that quantifies food waste in the U.S. MSW stream.

The materials flow methodology addresses problems associated with sampling, such as skewing due to atypical circumstances or local situations. Updates to materials flow models are relatively inexpensive once the analytical structure is in place (Bailie et al. 1999). A limitation of the methodology is that obtaining complete production data for every item discarded as solid waste is difficult (Bailie et al. 1999). Furthermore, the Franklin model is only applied to national-level data; USEPA guidance notes it may not apply to regional or state situations (USEPA 2013). Many of the methodology's assumptions are untested. For example, one iteration of the model assumed that the useful life of major household appliances was 20 years. This did not account for the substantial trade-in of older appliances to low income households, the prevalence of used parts being reused for appliance repairs (Rathje and Murphy 2001), or the occurrence of buying a replacement appliance but not disposing of the original.

Materials flow concepts are difficult to apply to food waste. Although USEPA's annual reports include estimates of the amount of food waste in the U.S. waste stream, the principles of the model cannot be applied to some organic wastes, such as food and yard waste (Tonjes and Greene 2012). Food waste is not generated by industrial processes where the kinds of materials used to create the materials are known and counted, the outputs are tracked, and product lifespans are understood (Nakamura and Kondo 2009). Therefore, data on food sales bear little relation to the generation of food waste because much food is consumed and considerable quantities of water may be added or removed from many food items between purchase and discard (Bailie et al. 1999). USEPA has acknowledged this, stating that 'quantities of MSW components such as food scraps and yard trimmings can only be estimated through sampling and weighing studies' (USEPA 2013). Because there is no detail on exactly which reports were used, their selection, or the process used to generate the food waste estimates, it is impossible to assess the assumptions, sampling error, or accuracy of these estimates.

6.3 Surveys and Interviews

Surveys and interviews involve asking participants direct questions about their food wastage in order to better understand food waste generation and disposal practices. This approach is beneficial in that it provides detailed information on why food is wasted and the specific types of food wasted, but is not as suitable for generating information on quantities generated. Okazaki et al. (2008) performed a survey of commercial food waste generators and managers in Hawaii and concluded that in-depth surveys were needed to ascertain food waste generation amounts, and that the results needed to be augmented by waste sorts.

Surveys produce subjective responses and require considerable investments of time and effort by respondents (Lebersorger and Schneider 2011). Furthermore, participants in household food surveys can be highly reactive (changing behavior during the survey period to avoid acknowledging food wasting) (Harrison et al. 1975, Griffin et al. 2009). The U.K.'s Waste and Resources Action Programme (WRAP) determined that most perceptions of disposal amounts were considerably less than reality (Quested and Johnson 2009). Rathje and Murphy (2001) also reported that when survey participants were asked to keep written records of the amount of food wasted, they rarely recorded food waste and, if they did, listed very small amounts. An issue with quantifying food waste with surveys is the culturally-driven moral implications of wasting food (Harrison et al. 1975), so that few people admit they unnecessarily waste food. In sum, people tend to greatly underestimate food wastage when self-reporting. Surveys may provide valuable qualitative information regarding wastage of food, such as the behaviors that lead to waste generation and disposal, awareness levels of food waste, motivators for waste reduction, and receptiveness to alternative management initiatives (Neff 2014). But, food waste needs to be quantified in a way which does not influence behavior of subjects and does not rely on self-reporting, so that waste sorts are needed to supplement survey data.

6.4 Food Diaries

Food diaries involve measuring and recording food wastage by waste generators themselves. An advantage of this approach is that food wasted through all methods of disposal (e.g., waste collection, sewer, home composting) can be identified together with possible reasons for the wastage. The diary method also enables the collection of background data on socio-demographics, behavior, and attitudes. Several food waste diary studies have been conducted (e.g., Langley et al. 2010, Koivupuro et al. 2012, Williams et al. 2012), but it is difficult to obtain large sample sizes (Koivupuro et al. 2012). A related technique involves researchers entering homes to weigh food waste. However, such intrusive observations present numerous sociopolitical obstacles (Rudd and Johnson 2008), which is one reason why these studies are seldom done.

Food diary studies require participants to expend considerable amounts of time (Langley et al. 2010), participants may forget or choose not to record some of their waste generated (Koivupuro et al. 2012), the results are subjective (Lebersorger and Schneider 2011), and participants may be reactive, altering their behaviors due to the study (Kantor et al. 1997). Rathje and Murphy (2001) found that when survey participants weighed their daily food waste, they greatly underestimated the amount of food waste actually generated by the household. Because people tend to feel bad or morally wrong about wasting food, their actions may be affected during the study (Williams et al. 2012), so it is not the best method for accurately quantifying food waste.

6.5 Food Supply Data

Some studies have calculated loss factors for various foods and applied them to overall food supply data to determine the amount of food loss and wastage. The USDA's Economic Research Service (ERS) generated estimates of food loss throughout the U.S. supply system by applying loss factors to the amount of food available for human consumption. This dataset contains spreadsheets for individual commodities with loss factors gathered from published studies and discussions with commodity experts (Kantor et al. 1997). The ERS data has been updated several times since 1997. Buzby et al. (2011) used the dataset to estimate the amount of food lost from the available food supply in the U.S. in 2008 and 2010 (Buzby et al. 2012). Hall et al. (2009) used food supply data (the FAO's food balance sheets which describe food supply by country) to quantify the energy content of food in the U.S., and combined this with a mathematical model of human energy expenditure to quantify food waste. The difference in food calories grown and those expended by people equals the food waste.

Loss factors may be understated or overstated due to limitations in underlying published studies that data are derived from (Kantor et al. 1997). Also, data are only available for several hundred individual food commodities by food group so some food commodities are missing, including products containing multiple foods (Buzby and Hyman 2012). The methodology does not assign food waste to management means, such as food disposals, animal feed, or the MSW management stream, so it provides little information on how much food waste needs to be managed.

7. Technologies and Polices for Food Waste Management

7.1 Background on Waste Systems

7.1.1 Objectives of waste management systems

Waste management systems can be broken down into six functional elements: waste generation, waste handling at the source, collection, transportation, processing, and disposal (Tchobanoglous 2009). A variety of technologies and policies are used throughout the waste management system to protect human and environmental health by reducing the negative impacts of waste and finding beneficial reuses for it. Being convenient and economical are also waste system priorities. Specific foci of waste systems will differ depending on the level of system sophistication.

Many developing countries still have unsophisticated, non-modernized waste systems, which has led to growing concern over the insufficiency of solid waste management there (Henry et al. 2006, Al-Khatib et al. 2007). Human health impacts result from open dumping and burning, as well as waste handling by unregulated and untrained informal sectors (Wilson et al. 2006, Cunningham et al. 2012). Public health tends to be the motivating factor in the development of waste policies in countries with unsophisticated waste management infrastructure, such as that in much of the developing world (Wilson 2007, Vergara and Tchobanoglous 2012). In the U.S. and Europe, public health was a key driver of waste practices from the 19th century through the 1960s, but now drivers have shifted towards environmental concerns. Because strict regulations exist and sophisticated waste management programs have been in place for decades, direct negative human health impacts of waste have generally been controlled and therefore, public health is no longer a major driver of waste policy (Wilson 2007). Changes in waste systems seek improved environmental protection through the optimization of waste management practices (that are economically viable), particularly those relating to energy consumption and climate change.

7.1.2 U.S. waste policy

Waste regulations set a framework for waste goals and harness local efforts towards common objectives. Municipal waste regulation in the U.S. is primarily governed under the Resource Conservation and Recovery Act (RCRA) (1976), which was the first comprehensive federal regulation containing guidelines for waste management and a legal basis for treatment, storage and disposal regulations (Constant 2002). It regulates daily operations of solid and hazardous waste management facilities and activities though a standards system and permitting (Ristau 2002). RCRA is structured to protect human health and the environment from potential hazards of waste disposal, conserve energy and resources, reduce the amount of waste generated, and ultimately ensure that wastes are managed in environmentally sound ways. Subtitle D specifically focuses on municipal waste practices that maximize the reuse of recoverable material and foster resource recovery (Pichtel 2005).

RCRA advocates for a waste hierarchy which defines the approaches and technologies that should be used preferentially (Pichtel 2005) through perceptions of their differential impacts on the environment. This hierarchy, which is generally recognized internationally as well, lists waste prevention as the first priority, followed by reuse, recycling, energy recovery, and landfilling as the final, least preferred option (Gertsakis and Lewis 2003, Manfredi and Pant 2013). Most waste system evaluation models examine systems in terms of the hierarchy with emphasis on reaching the top for system improvement (Coelho et al. 2012). Schmidt et al. (2007) demonstrate that the waste hierarchy is an appropriate principle guiding the handling of particular types of wastes using life cycle assessment. Much of the sustainability literature also supports the continuing relevance of the waste hierarchy as a guiding principle (Gertsakis and Lewis 2003).

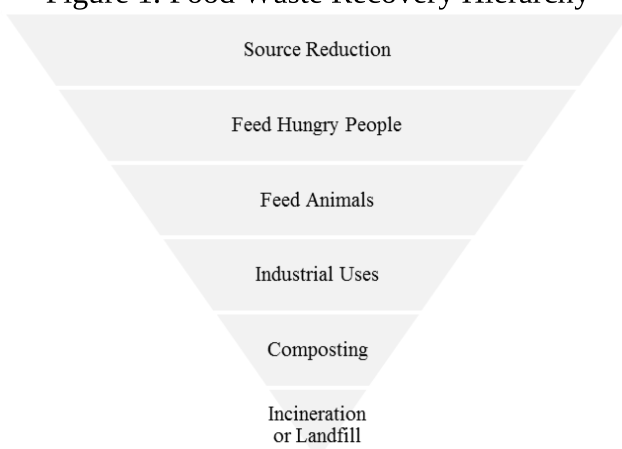
The hierarchy has been critiqued as lacking a scientific basis for ordering and its inability to be adapted to local situations. By some measures, the hierarchy may not lead to the identification of the most environmentally sound option for waste management (Manfredi and Pant 2013), so other integrated systems approaches to waste planning which account for site specific conditions have been proposed as alternatives (McDougall and White 2001). The European Union has suggested that deviations from the hierarchy are acceptable when justified by life cycle assessment, on the grounds that quantifying environmental impacts using a life cycle approach enables fair comparisons among alternative waste management options (Manfredi and Pant 2013).

Because solid waste management is implemented by state, regional, and local entities, RCRA’s Solid Waste program Title D encourages each state to develop comprehensive plans to manage nonhazardous industrial wastes and MSW, resulting in non-uniform waste programs and policies throughout the U.S. Most state and local programs have adopted policies aimed at diverting waste from landfills (Levis et al. 2010), including landfill disposal bans, such as for yard waste and electronics, volume based pricing system where customers are charged for waste collection services based on the amount of waste they throw away, deposit refunds for materials (generally known as bottle bills), and recycling mandates. Technology standards for treatment systems are major components of regulatory programs, and they generally aim to minimize environmental damages.

7.1.3 U.S. food waste hierarchy

USEPA has defined a food waste hierarchy specifying the preferred treatment for food waste (Figure 1), which assigns source reduction as the highest priority. The next preferable solution is to redirect food to feed people, followed by feeding it to animals. Otherwise, consideration should be given to using the food waste as a raw material for other industries, as a form of recycling. The next best option is to manage food waste in an environmentally sound way, preferably though composting. Another option at this stage is energy recovery through a technology such as anaerobic digestion. Finally, as a last resort, the food waste should be disposed through incineration or landfill (USEPA 2014a).

Figure 1. Food Waste Recovery Hierarchy



Adapted from USEPA 2014a

7.2 Technologies for Food Waste Recovery

Choices in waste technologies and management systems offer opportunities to reduce environmental impacts and produce beneficial end products, particularly through energy and materials recovery. Interest in diverting source separated food waste away from disposal has grown rapidly in the U.S. (Platt et al. 2014). The three primary treatments for organic wastes are landfilling, waste to energy incineration (WTE), and biological treatment. These approaches differ in their prime objectives. The primary purpose of landfilling and WTE is to manage wastes, while biological treatment aims to generate beneficial end products. In this way, biological treatment is like recycling where producing useful end products is the primary goal.

Aerobic composting and anaerobic digestion (AD) are the most common biological treatments for food waste. Large scale diversion of food wastes has yet to occur in the U.S., so most technology choices are largely prospective in nature. These are well-established, relatively frequently used technologies for treating organic wastes (McDougall and White 2001, Coker 2014). Both processes decompose organic materials; AD is entirely carried out by microorganisms. Inputs are organic feed stocks; outputs are compost and/or digestate and gases (in the case of AD, gases can be used to produce energy). Composting occurs in the presence of oxygen; AD is anaerobic. Composting is most efficient with a moisture content of 50 percent, while AD is most efficient under saturated conditions. There are benefits and drawbacks to each biological treatment process. AD and composting can be used in series to enhance the usefulness of the solid residues (Coker 2014). The ideal technology for a system likely depends on current operations, required yields, proximity of treatment facilities, political acceptability, public support, and financial situations (Lamb and Fountain 2010).

7.2.1 Composting

Composting is a natural biochemical process mediated by microorganisms in which organic materials, such as food and yard waste, decompose under aerobic conditions to form a rich, soil-like material. The process is essentially a batch process involving both micro and macro-organisms. First feedstocks are mixed, followed by degradation which produces heat; finally, the material is stabilized and no further biodegradation occurs (Coker 2014). Composting transforms organic feedstocks, reducing the volume and mass of waste by between 30 and 60 percent and changing complex organic substances into simpler humus-like solids. The remaining mass is released as carbon dioxide, water, and ammonia (Tchobanoglous and Kreith 2002). Composting as an element of carbon recycling in agriculture has been conducted by people for thousands of years (Fitzpatrick et al. 2005); more recently, large-scale, organized, centralized composting operations have been developed.

In the U.S., aerobic composting occurs at various scales and mechanisms, from simple backyard piles to expensive, advanced technologies. Large sites rely on technologies such as aerated (turned) windrow composting, aerated static piles, and in-vessel composting. Turned windrow composting involves placing organic wastes into long rows (windrows) and aerating them by turning the pile periodically through either manual or mechanical means. The piles are sized to allow for sufficient heat and temperature maintenance, and oxygen flow to the windrow's core. Aerated static pile composting, which was developed primarily for sewage sludge processing, involves forcing air through a trapezoidal compost pile, and agitation only when piles are combined or moved; these systems may be enclosed (typically in heavy duty bags) or outdoor. Enclosed windrow composting is essentially a hybrid between windrow composting and aerated static piles. It is performed in an enclosed area, typically a building, where forced air, aerated trenches, and pile agitation are used to accelerate the composting process (CCC, 2014). In in-vessel composting, organics are placed into an enclosed, rigid container (e.g., drum, silo) with controlled environmental conditions (temperature, moisture, aeration). Detailed descriptions of these technologies are given in Diaz et al. (2007) and Platt et al. (2014).

The composting process is largely controlled by the ratio of carbon to nitrogen (C:N ratio). A C:N ratio of 25:1 or 30:1 maximizes the composting rate while minimizing odors. Higher ratios lead to slower composting rates, while lower ratios tend to result in odor generation. The C:N ratios for several compostable materials commonly found in MSW are given in Table 11.

Table 11. Representative C:N Ratios of Compostable Components of MSW

Waste Category	**C:N Ratio**
Yard Waste	30:1
Grass clippings	15:1
Leaves	60:1
Other yard waste	30:1
Food waste	15:1
Paper	120:1

Adapted from Bailie et al. 1999

Composting reduces the volume and mass of materials, kills pathogenic microorganisms, and creates a stable organic product (compost). Finished compost is used primarily to introduce carbon to soils or as a potting medium. It is also used to control erosion, as mulch, and to engineer soil. It improves physical properties of soil, and increases water retention and the availability of essential nutrients (Hargreaves et al. 2008).

Yard waste composting has generally been successful in the U.S. operationally and financially. This success and the overall public acceptance of source-separated yard waste composting programs has created momentum to expand feedstocks to include food waste (O'Brien 2012). Food waste composting is different in many ways from yard waste composting as food waste is less homogenous, tends to be more odorous, and is more likely to attract vectors. Most food waste composting operations handle both food and yard wastes together (Levis et al. 2010). Co-composting is attractive because food waste has a low carbon to nitrogen ratio (Table 11), which makes it difficult to compost on its own. Combining food waste with yard waste brings the C:N ratio closer to 25:1 (Bailie et al. 1999). Small quantities of food wastes have been incorporated into unenclosed yard waste windrow operations without too many odor or vector reported issues, although these issues would increase with large amounts of food waste feedstock. Isolated composting facilities may be able to compost high quantities of food waste outdoors; larger scale or urban-suburban food waste management will likely require enclosed technologies.

7.2.2 Anaerobic digestion

Anaerobic digestion (AD) is a microbe-mediated process involving the degradation of organic materials in anaerobic environments. Digestion is somewhat more complex than composting, as the organic material is broken down through the sequential steps of hydrolysis, acidogenesis, acetogenesis, and methanogenesis (Coker 2014). The process yields biogas, which is approximately 60 percent methane and 40 percent carbon dioxide; the biogas can be recovered and treated so it may be used for energy production (USEPA 2014c). Often, the AD process is followed by composting which produces compost from the nutrient rich digestate. Thus, AD can produce energy and compost (if digestate is treated after the initial AD process); this stabilization of organic matter reduces potential environmental impacts from air and water pollutants and it reduces the volume and mass of waste (Rapport et al. 2008). Because AD occurs in closed, controlled settings, produced gases can be captured, thus preventing their release to the environment. AD technologies are generally classified based on four criteria: (1) solid content of feed; (2) number of stages; (3) operating temperature; and (4) means by which feed is introduced into the anaerobic reactor (Levis et al. 2010). Rilling (2005) has described AD technologies in detail.

AD is attracting more attention for food waste management (Zhang et al. 2014). In July 2009, a Bay Area, CA utility company became the first facility in the U.S. to convert postconsumer food waste to energy using AD. The U.S. wastewater treatment industry uses AD to process and treat sludge (Rapport et al. 2008). In 2014, there were 1,500 operating anaerobic digesters at wastewater treatment facilities; about 250 use the generated biogas to produce energy (the remainder flare the gas) (American Biogas Council 2014). There are 230 anaerobic digesters at dairy, swine and poultry operations (USEPA 2014c). These treat manures to mitigate some of the impacts associates with large-scale animal farming and produce energy (Rapport et al. 2008). There are a few pilot scale operations in the U.S. that add food waste to the feedstocks. Since AD is a saturated process, the relatively high moisture content of food waste is more suitable for treatment and energy recovery compared to thermochemical conversions, such as combustion where water is a process impediment (Zhang et al. 2014, Zang et al. 2007).

7.2.3 Incineration

Waste-to-Energy (WTE) incineration is the conversion of waste material to gas products and solid residues by the controlled burning of wastes at high temperatures (Bailie et al. 1999), leading to the production of energy. Mass-burn technology is used, meaning wastes require little-to-no pre-processing except for removal of bulky wastes, and result in near complete combustion. Process outputs include heat and energy, ash, and stack emissions, such as acidic gases (carbon dioxide, carbon monoxide, oxides of sulfur and nitrogen), persistent organic compounds (dioxins and furans), heavy metals (cadmium, thallium, mercury, antimony, arsenic, chromium, cobalt, copper, manganese, nickel, lead, and vanadium), and particulates (Crowley et al. 2003). Incinerating food waste allows the energy within the food to be used for electricity generation, but no nutrients are recovered. The high moisture content of yard and food wastes (about 65 percent moisture) leads to a lower heating value than other materials, such as plastics (Bailie et al. 1999). Waste materials are burned at temperatures high enough to consume nearly all combustible materials, leaving only ash and noncombustible materials, such as metals, to be recovered or disposed in a landfill. Ash generation is about 25 to 30 percent of infeed tonnages, and about 10 percent of infeed volumes.

Modern WTE facilities have extensive air pollution control equipment, and emissions have been reduced considerably since the introduction of incineration. In the early 20th century U.S., in-house incinerators were common resulting in a high ash fraction in American trash. Later, large scale, industrial incinerators were opened, but significant growth was limited by pollution concerns. RCRA (1976) and the Clean Air Act (1990) set strict standards for pollution from incinerators, which led to the development of air pollution control technologies (Louis 2004) which greatly minimize furan, dioxin and mercury emissions to the environment. WTE has been applied widely to generate energy from waste materials, substantially reduce waste volume, and destroy harmful materials (chemicals and pathogens).

7.2.4 Sanitary landfilling

Sanitary landfills are landfills that minimize nuisances or hazards to public health or safety. Modern sanitary landfills have leachate collection and treatment systems, landfill gas controls, and environmental monitoring systems (Tchobanoglous 2009). Their locations are carefully selected to avoid environmentally sensitive areas, and waste is covered with a layer of compacted cover (usually soil) at the end of each operation day for odor and vector control. In the anaerobic landfilling environment, food decomposes producing methane gas. This gas may be captured and used for energy if the landfill has proper landfill gas capture technologies. Landfill gas collection efficiencies have been estimated to be between 50 and 100 percent depending on cover type and the extent of collection system coverage (Barlaz et al. 2009). There are 636 operational landfill gas to energy projects in the U.S. (out of the approximately 2,400 currently operating or recently closed landfills). Those without capture or flaring allow methane to be released to the environment, causing environmental damage as methane is a more potent greenhouse gas than carbon dioxide (USEPA 2014e).

7.2.5 Other technologies

Food waste has been used as animal feed, although not all food waste is suitable for all animals, especially livestock. Laws in the U.S., Australia, and Europe prevent most food waste from being fed to animals due to the risk of meat contamination, such as with Bovine Spongiform Encephalopathy (Mad Cow Disease). In the U.S., if the wastes do not contain meat or other animal parts, there are no federal restrictions, although some state laws regulate feed. If the food waste contains meat or animal parts, or has come into contact with meat or animal products, it can be used for pig feed under the Federal Swine Health Protection Act, which requires the waste to be boiled first (USDA 2009). Specific state regulations concerning feed vary from strict bans to regulations regarding applicable materials which may be used, or handling and treatment processes. Food waste may also be pelletized for aquaculture food (Mo et al. 2014).

There are several new technologies being developed for food waste management that are not yet fully mature. One such technology, hydrothermal carbonization, involves a thermal conversion process that has been shown to be feasible for the conversion of wet feedstocks, such as food waste, to high energy, high carbon-containing solid residue, hydro-char (Berge et al. 2011). Bioreactor landfills are an alternative management strategy for food waste if it is not source separated and diverted, but rather commingled with all MSW. Unlike traditional sanitary landfills, bioreactor landfills are operated in ways to enhance degradation processes. This is usually accomplished by increasing the liquid content of the fill, primarily by not removing leachates and instead re-circulating them through the wastes. Enhanced degradation increases the generation of methane, which means there is greater recovery of the inherent energy in the wastes. The necessary assumption is there will be effective methane capture and the subsequent production of electricity or liquid fuels (Warith 2002). Bioreactor landfills are still novel technologies and are mainly in the development and testing phase (USEPA 2014d). Other novel technologies for MSW include pyrolysis, gasification, and hydrolysis, although these technologies are not market ready for wide scale deployment.

7.3 Policy Mechanisms for Food Waste Management

There are several alternative frameworks guiding waste policy which differ in their primary targets, although they all seek to achieve maximal benefit from waste and minimize damages. Frameworks include zero waste initiatives (no waste landfilled or incinerated), achievement of maximal recovery/recycling, adherence to the waste hierarchy, or no landfilling of wastes. Food waste recovery may be an integral component of programs under any of these frameworks, although the immediate objectives for targeting food waste may differ (e.g., increase recycling rate, decrease disposal rate). There are various specific policy mechanisms which may be leveraged under any of the frameworks to reduce or divert food waste.

A policy option for food waste management is a ban which makes it illegal to dispose (landfill or incineration) food waste. The first such bans were enacted in cities, including Seattle, WA and San Francisco, CA, and have also been implemented in states and provinces, such as

Nova Scotia, Canada, and Massachusetts. These regulations are modeled after yard waste landfill bans, commonly adopted throughout the U.S. in the 1990s in over 20 states (Platt and Goldstein 2014). The success of yard waste disposal bans, as measured through wide implementation and a reduction of disposed yard waste, suggest that food waste bans may also minimize food waste disposal.

Some bans prevent disposal though landfilling only, such as that implemented in the EU. The EU passed the Landfill Directive in 1999, requiring the biodegradable portion of MSW to be reduced compared to 1995 levels: 25 percent within five years (2006), 50 percent within eight years (2009), and 65 percent within 15 years (2016) (European Union 1999). By targeting landfilled biodegradable waste, the Directive aimed to promote options to increase and improve recycling and recovery, and reduce greenhouse gas (GHG) emissions.

Pay-as-you-throw (PAYT) has been shown to generally reduce the disposal of MSW, and the effect might include food waste reductions or increased recovery. These volume or weight based pricing systems for waste (recyclable and compostable materials are free) have been documented to reduce disposal tonnages by 10 to 20 percent (Skumatz 2008). With PAYT, residences can save money by disposing of fewer materials, thus encouraging them to source separate organics and recyclables for alternative treatment, or to reduce generation.

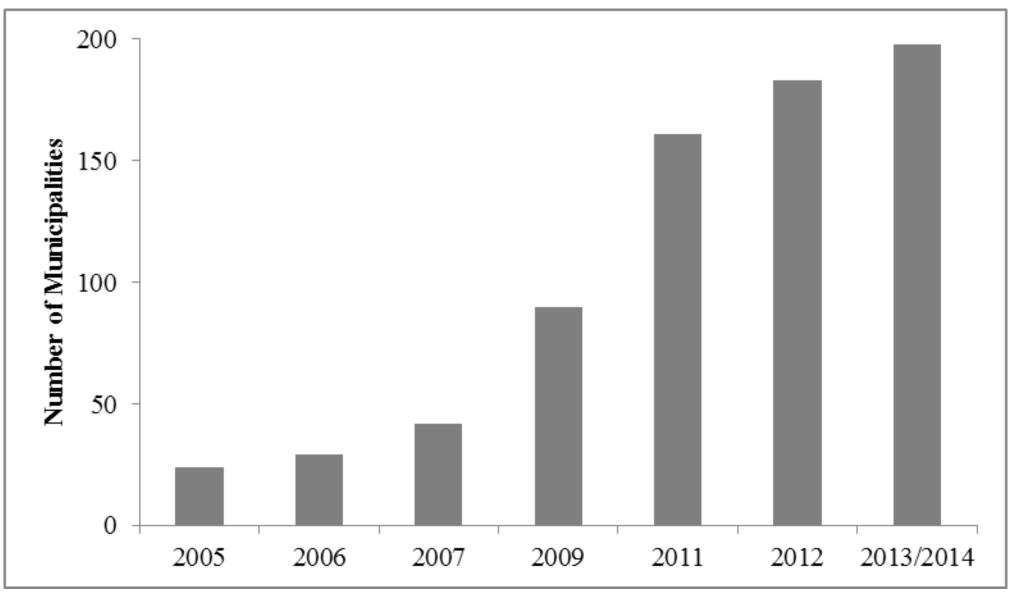
Another mechanism to reduce food waste or encourage diversion is to limit the frequency of trash collection which can reduce disposal and increase recovery. Portland, OR, for example, switched to every other week trash collection with weekly recycling and organics service in 2011. Lamb and Fountain (2010) found that separate food waste collection is unlikely to be economically justified without every other week trash collection. Reducing trash collection allows the excess costs associated with implementing a food diversion program to be offset.

One other possible driver for reducing or diverting food waste is regulatory requirements mandating that a specific proportion of waste be diverted from the waste stream or recovered. One such policy was imposed in California (AB939) where the state set a goal of 25 percent diversion by 1995, and 50 percent diversion by 2000. Problematically, some of this legislation seems to be passed without critical inputs regarding whether specific goals are attainable in the timeframes being mandated. Therefore, it is essential to ensure that mandated goals are realistic and achievable.

8. Current State of Food Waste Recovery

Interest in diverting source separated food wastes from disposal is growing rapidly in the U.S., and is reflected in both federal and state policies (Platt et al. 2014). U.S. governments have begun to develop policies and regulations that specifically focus on food waste in MSW. Some programs are aimed directly at commercial establishments, while others focus on the residential stream, and some target all food waste in MSW (institutional, residential, and commercial). Institutions, such as universities, are targeting food waste on their own. BioCycle magazine (2015) found 198 communities in 19 states offered residential food waste collection in 2013 and 2014, an increase from prior years (Figure 2). Most residential programs in the U.S. are located on the west coast in Washington (57 programs) and California (65 programs). BioCycle estimated 2012 residential food waste collection programs reached 2.74 million households (Yepsen 2015).

Figure 2. Number of U.S. Municipalities with Source Separated Food Waste Collection



Adapted from Yepsen 2015

8.1 U.S. Federal Level Policy

The USEPA actively promotes its food waste hierarchy (discussed in section 7.2). In June 2013, USEPA and USDA announced the U.S. Food Waste Challenge, a collaboration to raise awareness of the environmental, health, and nutrition issues created by food waste. The program encourages businesses and institutions to prevent, donate, or recycle food waste. Another challenge, the USEPA's U.S. Food Recovery Challenge (USDA 2014c) encourages participants from across the food chain to list food waste management activities to allow for the dissemination of information about best practices and stimulate the development of increased food waste reduction and management programs. USEPA requires participants to set specific quantitative food waste goals, and works with them to measure progress and attain goals. These two federal initiatives have been effective at increasing public awareness of the amount of food waste generated in the U.S., as well as encouraging food waste generators to minimize food waste and divert it away from disposal (Platt et al. 2014).

Some institutions, such as universities, schools, and hospitals, are diverting food waste under these federal challenges. In 2014, 154 universities in the U.S. were members of the USEPA Food Recovery Challenge Program and had source separation and alternative management program for food waste. Other business and institutions which take part in the Food Recovery Challenge include resorts, hotels, police departments, and supermarkets.

8.2 U.S. State Level Policy

Policies have been implemented at the state level to encourage or mandate diversion of food waste (Table 12) (Platt et al. 2014). Vermont was the first state in the U.S. to require mandatory recycling and composting of all food waste (even from residential sources) by 2020. Other New England states have followed. Maryland introduced a bill in 2014 to ban large-scale generators of food waste from disposing food waste, but the legislation failed to pass (Platt et al. 2014). California has a regulation requiring commercial generators to separate food and yard wastes and to arrange for recycling service beginning in 2015. Other states promote food waste management, although it is not part of a mandated policy. New York calls for planning units to evaluate and implement, to the greatest extent possible, recovery of materials, including food scraps (NYSDEC 2010). Washington's Beyond Waste Plan calls for eliminating wastes wherever possible, and using remaining wastes as resources (WA State, 2015).

Table 12. Current State-Level Food Waste Disposal Bans

State	Year Enacted	Whom does the law apply to?	Details	Date Provisions
California (AB 1826)	2014	Commercial generators of more than 8 cubic yards of food or yard waste per week	Food & yard waste must be source separated and recycled	To be implemented in 2016; by 2017 generators of more than 4 cubic yards per week must comply
Connecticut (Public Act No.11-217 amended by Section 4 of P.A. 13-285)	2011	Commercial food wholesaler or distributor, industrial food manufacturer or processor, supermarket, resort or conference center, located less than 20 miles from an authorized source-separated organic material composting facility, & generates at least 104 tons per year of organic materials	Organics must be separated from other solid waste & recycled at any authorized composting facility has available capacity and will accept such material	Applied January, 2014; in 2020, tonnage limit=52 tons per year
Massachusetts (310 CMR 19.000)	2014	Commercial & institutional generators of 1 ton or more of organics per week	Organics are banned from disposal & must be donated, re-purposed, composted, used as animal feed or sent to AD	Applied October, 2014
Rhode Island (H7033 SubA)	2014	All generators of 104 tons or more of organics per year within 15 miles of a certified processing facility	Must divert organics from landfills	To be implemented in 2016
Vermont (Act 148)	2012	All waste generators of 104 tons or more of food residuals per year within 20 miles of a certified processing facility	Food residuals must be source separated & delivered to a location that manages food waste; haulers must offer curbside collection 2017	Applied 2014; annual tonnage requirements decrease with time (2015: 52 tons; 2016: 26 tons; 2017: 18 tons; 2020: any person generating any amount of food waste [no longer a provision for distance])

8.3 U.S. City Level Policy

San Francisco, CA instituted the first local municipal ordinance in the U.S. to require source separation of all organic material in 2009 (San Francisco Department of Environment, 2014). The City enacted a complete ban on food waste disposal in landfills for all generators which requires residents to separate their recyclables, compostables, and landfill trash. Seattle, WA offers a similar ban which is in its early stages of implementation, and currently offers a voluntary curbside food waste collection (Seattle Public Utilities, 2015). San Diego collects food waste from 33 commercial and institutional establishments by the City’s franchised haulers and it is composted in open windrows at the City’s composting facility (City of San Diego 2014).

The program accepts food scraps, coffee grounds, parchment paper, and paper towels and napkins from kitchens. New York City passed legislation (Commercial Organics Law) in December 2013 requiring commercial food scraps from the largest food service establishments and other commercial operations (e.g. stadiums, chain restaurants) that generate significant amounts of food waste to be recycled by July 1, 2015. The regulation requires food waste to be sent to a composting or AD facility or a transfer station that delivers to such a facility. New York City also has several pilot food waste collection routes for residential food waste. Other cities which have adopted food waste source separation include: Arvin, CA; Los Angeles, CA; Palo Alto, CA; Boulder, CO; Loisville, CO; Denver, CO; Lexington, KY; Cambridge, MA; Ann Arbor, MI; Hutchinson, MN; Ithaca, NY; Huron, OH; Portland, OR; Austin, TX; San Antonio, TX; Brattleboro, VT; Olympia, WA; Fitchburg, WI; and Madison, WI. Counties with programs include: Alameda, CA; Santa Cruz, CA; San Mateo, CA; Boulder, CO; Howard, MD; Hennepin, MN; Marion, OR; and King, WA.

8.4 Current Facilities Accepting Food Waste in the U.S.

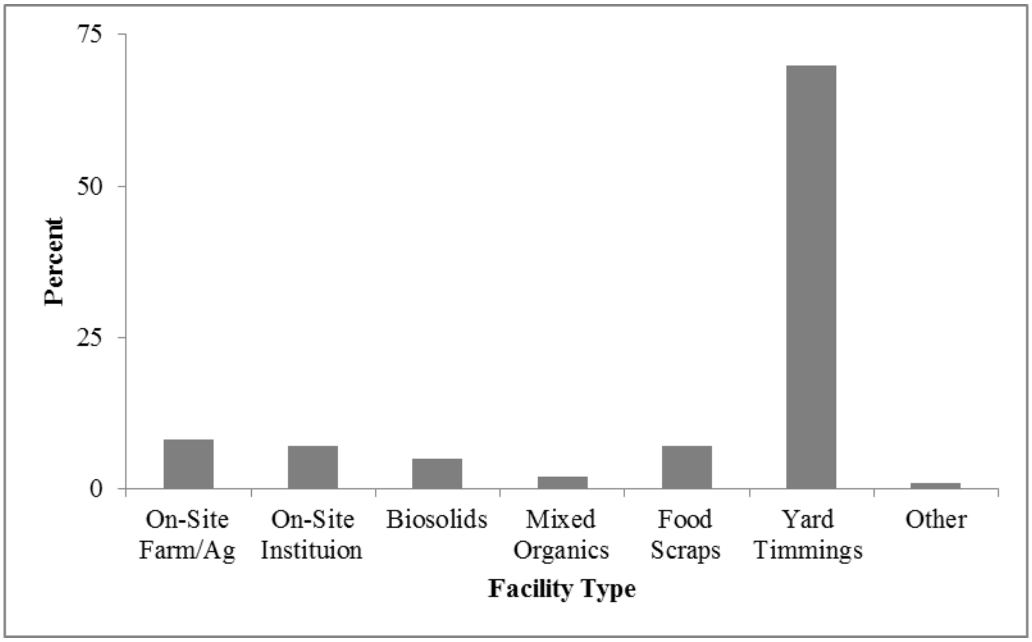
8.4.1 Composting facilities

About seven percent of U.S. composting facilities accept food scraps (347 of 4,914). Most food waste composting facilities co-compost with yard trimmings from municipalities, commercial landscapers, and homeowners; few are dedicated to food waste (Platt et al. 2014). Figure 3 details the yard waste and food waste composting facilities in each state. New York has the highest number (45) of sites accepting food waste, followed by Washington (29), Massachusetts (27), California (26), Pennsylvania (25) and Illinois (21) (Table 13) (Platt et al. 2014). In 2010, Levis et al. (2010) found that about 80 percent of food waste composting facilities accept less than 5,000 metric tons of food waste per year, and over 90 percent manage less than 50,000 metric tons of food waste per year. At that time, half of the food waste composting facilities were commercial or municipal facilities, while the remainder was at universities and farms. Only one quarter of food waste composting facilities accepted residential food waste (Levis et al. 2010).

A website was launched by BioCycle magazine and the Biodegradable Products Institute in 2007 (FindAComposter.com) to facilitate composting by listing facilities that accept food waste. The free, publically searchable database listed 264 food waste composters as of

December 2013. BioCycle magazine found an additional 295 sites that were permitted to accept food scraps, but were not listed (Krossovitch et al. 2014).

Figure 3. U.S. Composting Facilities by Type



Adapted from Platt et al. 2014

Table 13. U.S. Composting Operations Targeting Yard and/or Food Waste

State [a]	Accepting Yard Waste	Accepting Food Waste
Alaska	0	0
Arizona	4	
Arkansas	20	1
California	48	26
Colorado	2	2
Connecticut	109	3
Delaware	0	2
Florida	257	2
Georgia	1	1
Idaho	7	4
Illinois	42	21
Indiana	119	11
Iowa	86	7
Kansas	103	11
Kentucky	35	
Maine	52	10
Maryland	7	4
Massachusetts	221	27
Michigan	119	7
Minnesota	129	9

Mississippi	9	3
Missouri	18	6
Montana	30	1
Nebraska	10	0
New Hampshire		9
New Jersey	295	1
New Mexico	16	
New York	329	45
North Carolina	16	7
North Dakota	43	0
Ohio	299	20
Oregon	44	10
Pennsylvania	350	25
Rhode Island	22	3
South Carolina	107	1
South Dakota	146	0
Tennessee	3	2
Texas	33	4
Utah	18	4
Vermont	1	13
Virginia	8	1
Washington	45	29
Wisconsin	225	14
Wyoming	25	1
Total	3,454	347

Adapted from Platt et al. 2014

[a] Based on the states that participated in the *BioCycle* survey. All but six states (Alabama, Hawaii, Louisiana, Nevada, Oklahoma, West Virginia) participated. The blank cells indicate data were not provided.

8.4.2 Anaerobic digestion facilities

AD facilities processing food waste in the U.S. are fewer in number compared to composting facilities. In 2008, there were no commercial-scale solid waste digesters operating in the U.S. (Rapport et al. 2008); by 2010 one medium-scale facility was accepting source separated organics in North America, which was located in Canada (Levis et al. 2010). A 2013 study found six wastewater treatment facilities with AD that accept food waste, nine dairy farms with AD that accept food waste, and seven operational or under-construction, standalone AD facilities for food waste in New England alone (Fitzgerald 2013). However, most of the standalone facilities were industrial scale food processing centers, such as breweries, and did not accept municipal wastes. Additionally in 2014, an agreement was signed between a large organic fertilizer firm (NEO Energy) and a biogas firm (Himark BioGas International) to design, build, and commission three AD and fertilizer plants to process farm and food waste in Massachusetts and Rhode Island. In Massachusetts, several million dollars have been provided by the state in the form of low interest loans for those building AD facilities to help address high capital and annual costs of AD (Whyte and Perry 2001). Two AD facilities for food waste have also been planned in New York State (in Ithaca and Yaphank).

8.5 Overview of Food Waste Recovery Outside the U.S.

English language literature and documents concerning food waste management in North American and Europe are generally available, but there is a distinct lack of food waste related publishing (in English) for the rest of the world, particularly South America and Africa (Schneider 2013a). The following details some food waste recovery programs outside the U.S. based on available information.

8.5.1 Europe

In 2014, the European Parliament adopted a non-legislative resolution, the European Resolution on Food Waste Avoidance, calling for action to reduce food waste 50 percent by 2025 through a coordinated strategy combining EU and Member State measures to improve the efficiency of food supply and consumption in all stages of the food supply chain. Specific measures suggested include awareness campaigns informing the public how to avoid wasting food, the development of school courses explaining best practices for storing, cooking and disposing of food, and the adoption of a dual-date labelling system showing when food may be sold until compared to when it should be consumed. The Parliament designated 2014 as the 'European year against food waste' (Jakubov and Buondadonna 2012). The European Commission funded a four year program directed at food waste, FUSIONS (Food Use of Social Innovation by Optimizing Waste Prevention Strategies), which aims to standardize the measurement of food waste and start initiatives to reduce it (EU FUSIONS 2014).

Solid waste treatment by AD is common in Europe and is used widely for broader ranges of materials than in the U.S. (Levis and Barlaz 2011a). There are more AD treatment facilities for MSW in Europe because the EU Landfill Directive prevents landfilling of biodegradable wastes, and the relative scarcity of landfill space has led to high tipping fees at landfills (EU tipping fees are about double those in the U.S.) (O'Brien 2012, EEA 2013). Also, because energy costs are higher in Europe, renewable energy tariffs support AD facilities, and utilities are often required to connect renewable energy producers to the grid. In 1999 European AD plants processed about one million tons per year of mixed MSW or source separated organics in 53 plants. Globally, in 2005, it was estimated that there were 74 AD facilities in operation (mostly in Europe) processing organics or mixed MSW. In 2006, it was estimated that the number of commercially operating (or under construction) AD facilities worldwide had increased to 124, processing almost four million tons per year of waste (Kelleher 2007). In 2008, Europe had over 160 major AD facilities that collectively processed more than 11,000 tons per day of MSW organics (Rapport et al. 2008). A 2014 report in the U.K. indicated that there were 51 local authorities in the U.K. that used AD for organic waste treatment, double the amount of municipalities using it in 2013 (WRAP 2014).

Separate collection of household food wastes have been instituted in Austria, Germany, Sweden, U.K., and the Netherlands (European Compost Network 2014). Other European countries, including Norway, Belgium, Switzerland, are also working at source separating food waste (Kidby 2014). Policies in individual countries include a Swedish program for separate collection of food waste from households, restaurants, and supermarkets, targeting 50 percent of the food waste to separately collected and biologically treated by 2018 (Schott et al. 2013). Their program resulted in about 60 percent of Swedish municipalities employing food waste collection programs; an additional 70 are in the planning stage. Typically household waste is separated into two bins (one for food waste and one for combustible waste). Malmo has incorporated an innovative food waste management system into its Western Harbor region, using a vacuum system specifically designed for collecting food waste. Residents collect food waste in paper bags (provided by the municipality), then throw the bags into inlets where they are transported in pipelines to refrigerated storage units. The units are later emptied into trucks and the wastes are managed by AD.

The U.K.WRAP (Waste and Resources Action Programme) has long supported food waste reduction through its ‘Love Food, Hate Waste’ campaign beginning in 2007 to address avoidable residential food waste. WRAP has conducted research to quantify food waste in the U.K and causes for this wastage (WRAP 2011, WRAP 2013). WRAP quantified growing food waste collection efforts, which doubled from 2008 to 2009. By 2011, 47 percent of waste programs provided household collection services, and by 2012, 11 percent of all food waste was source separated for treatment by composting or AD (WRAP 2013).

8.5.2 Other parts of the world

Both South Korea and Taiwan have banned the disposal of food waste in landfills. In 2005, South Korea made it illegal to landfill food waste, and all municipalities of 100,000 or more had implemented food waste recycling. It is mandatory to separate food waste into special bags which must be purchased, thus making generation of more food waste more costly. Taiwan implemented a Food Recovery and Reuse Plan in 2001 to promote food waste source separation and recovery; about three-quarters of the collected food waste is used for pig feed, and the rest is composted. Japan passed a Food Waste Recycling Law in 2001 which targets large waste generators (more than 100 tons of food waste generated per year), requiring them to recycle their food waste through composting, animal feed, or incineration. The law called for 20 percent reduction of food waste by 2006. In 2007, the law was revised to promote a recycling loop, requiring food industries to purchase farm products that are grown using food waste derived compost/animal feed and businesses had to recover 66 percent of food waste by 2012 (Takata et al. 2012). This law only targets large scale generators; little household food waste in Japan is currently recycled, and few municipalities have food waste recovery programs in place.

9. Conclusion

It is likely that fundamental changes in the U.S. food systems brought on by accelerated industrialization, globalization, and economic growth over the past 25 years have altered food waste disposal practices. Specific factors include an increase in the disconnection from agriculture and food sources, a decrease in consumer concern about food waste, and a devaluating of food. Additionally, food waste disposal may result from over stocking and over preparation of food (Quested and Johnson 2009), confusion over food labels (Kosa et al. 2007), increased portion sizes (Young and Nestle 2002), or misconceptions regarding food safety (Pearson et al. 2013). The implications of food wastage, particularly its social, economic, and environmental effects, have led to increased concerns over it (Nixon 2015). Efforts to prevent or manage food waste better have recently been proposed, several states have implemented disposal bans, and the number of biological treatment facilities accepting food waste has increased throughout the U.S.

Chapter 3. Quantification of Food Waste Disposal in the United States: A Meta-Analysis

1. Introduction

Although food waste has been identified as a significant social, nutritional, economic, and environmental problem (Sobal and Nelson 2003, Pearson et al. 2013), considerable uncertainties remain regarding food waste quantities in the U.S. and globally (Lebersorger and Schneider 2011). There is a scarcity of data on food waste all throughout North America (Kelleher and Robins 2013), Europe (Brautigam et al. 2014), and the rest of the world (Parfitt et al. 2010), and the data that do exist tend to be incomplete and outdated (Cuellar and Webber 2010). Few peer-reviewed or major studies estimating quantities of food waste have been conducted (Buzby and Hyman 2012), and those that have been done utilize different methodologies (Gjerris and Gaiani 2013), making it difficult to compare findings across studies or aggregate findings. These data gaps have led to recent calls for further research on wasted and lost food (Gustavsson et al. 2011, Brautigam et al. 2014). Oelofse and Nahman (2013) concluded that continued research on food waste management is essential, especially to generate accurate data on waste quantities and composition.

1.1 Study Objectives

Because food waste is a major component of municipal solid waste (MSW) and is increasingly targeted for separate management, a detailed understanding of food waste disposal quantities would be useful. Multiple recent waste characterization studies in the U.S. have indicated large quantities of food waste in the MSW stream (wastes from residential, commercial, and institutional sectors), although these studies have not previously been collated or statistically analyzed. The primary study objective here was to utilize this extensive dataset of waste characterization sort studies to quantify disposed food waste in a transparent, repeatable, and systematic way, and to determine if specific factors drive increased disposal. The powerful statistical and conceptual tools of systematic review and meta-analysis were used as a strong alternative to the often more obscure methods used to estimate food waste to date. This study focused on food waste in MSW, which makes it an expansion beyond more common analyses which look only at general food losses and waste, or wastage from specific generator types, such as food services (Whitehair et al. 2013) or grocery stores (Freeborne 1993).

Two other studies have collated waste characterization studies, although the specific methods, scales, and overall objectives differ considerably from this work. The U.K.'s Waste and Resources Action Programme (WRAP) (2011) collated and analyzed data from waste composition studies in the U.K. that focused on disposed food waste. The waste characterization collation findings for households were averaged and combined with estimated disposal tonnages to generate overall disposed food waste quantities for 89 local authorities in the U.K. Staley and Barlaz (2009) combined 11 state waste sorts using the sample arithmetic mean to create an approximation of the wastes discarded in landfills. The data were used to estimate landfill gas emissions that would result from particular organic wastes.

Examining the amount of waste that is currently being disposed shows the amount of waste that has yet to be recovered from the disposal stream, thus indicating how much waste is available for prevention or alternative treatments. Recycling programs are well-established and usually mandatory, so it is reasonable to assume these efforts will continue. Analyzing materials that are still being disposed defines areas where improvement can be achieved. A better understanding of the MSW stream also allows for improvements to key inputs for waste models, such as life cycle assessments (LCA), and better, data-driven, policy development and decisions.

A second objective was to examine the United States Environmental Protection Agency's (USEPA) estimates of food waste disposal and determine their consistency with waste characterization study data. There are few alternatives to the USEPA nation-wide characterization of MSW and USEPA waste characterization data are overwhelmingly used when discussing U.S. MSW (e.g., Zhang et al. 2007, Saer et al. 2013). The USEPA approach has been criticized as being inappropriate to characterize food waste generation (Tonjes and Greene, 2012) and waste characterization sort results have been found to be different from USEPA estimates (ESDI 2004, R.W. Beck 2005, Abramowitz and Sun 2012). The definitive data generated here can serve as a compelling test of the accuracy and applicability of the long-standing USEPA dataset.

1.2 Methods for Quantifying Food Waste in MSW

Two methods for waste characterization and quantification of MSW are waste characterization sorts and materials flow models. Waste characterization sorts, the most common quantitative method to analyze waste streams, involve the representative sampling, sorting, and weighing of wastes originating in a target waste shed to determine the proportion of waste types in the samples (Bobman and Culbertson 2010). They tend to be applied to the whole disposed MSW stream, and exclude wastes disposed outside of the MSW system, such as industrial food waste or wastes fed to animals. There are numerous characterization studies that have been completed in the U.S., thus creating an extensive dataset. One third of states have conducted characterization studies and there are many city or county level studies (Bobman and Culbertson 2010). Most studies follow the general principles of the ASTM method for waste characterization (ASTM D 5231-08), which helps make results between studies comparable; these studies have also been assessed as consistent and reliable (ASTM 2008, Staley and Barlaz 2009, Bobman and Culbertson 2010, Lebersorger and Schneider 2011).

MSW is also characterized and quantified through materials flow modeling, which is performed on a national scale by Franklin Associates, Ltd. under contract with the USEPA. Waste generation is determined by making specific adjustments to industrial production data, such as for imports/exports and product life spans. The model breaks waste estimates into specific waste categories and estimates how much generated waste is treated by specific approaches, such as recycling, landfilling, or composting. The USEPA currently publishes the findings in a Facts and Figures Report (USEPA 2013), which is the only large scale, country-wide study that describes the quantities of food from residential, institutional, and commercial sources in the U.S. MSW stream. Advantages of this methodology are that it aims to quantify entire waste streams instead of relying on sampling, and updates to the models are relatively inexpensive once the analytical structure is in place (Bailie et al. 1999).

Although the USEPA annual reports quantify the amount of food waste generated and disposed in the U.S., materials flow concepts are inappropriate for food waste (Tonjes and Greene 2012). Food waste is not generated by industrial processes where the kinds of materials used to create the materials are known and counted, the outputs are tracked, and product lifespans are understood (Nakamura and Kondo 2009). Therefore, data on food sales bear little relation to the generation of food waste because much food is consumed (Bailie et al. 1999). USEPA has acknowledged this, stating that 'quantities of MSW components such as food scraps and yard trimmings can only be estimated through sampling and weighing studies' (USEPA 2013). The earliest iterations of the USEPA estimations relied on one or two site-specific sampling efforts (Smith 1975). More recently, USEPA states that food scraps, yard trimmings, and a small amount of miscellaneous inorganic wastes are accounted for by compiling data from a variety of waste sampling studies in combination with demographic data on population, grocery store sales, restaurant sales, number of employees, and number of prisoners, students, and patients in institutions.

In the 2011 report, USEPA stated that 17 residential food waste measurement studies provided the basis for the average per capita food waste disposal (USEPA 2013). There is no detail provided on exactly which reports were included, the way they were selected, or specifically how they were used to generate the food waste estimate, so it is impossible to assess the assumptions, sampling error, or accuracy of these estimates. In the same report, food waste generation is documented by 70 references. Besides the eight personal communications dated 2010, only seven of the references were from 2010 or 2011, none of which were waste characterization studies (rather, they were labor statistics, news articles, general reports, or census data), indicating that sort data were not from current research. Furthermore, since USEPA updates estimates annually, it would be expected that data would also be updated through more recent references. When examining the reference list, it is not clear which studies USEPA considered to be the 'residential food waste management studies' that provided the basis for the food waste estimates. No other details were given, implying that a systematic or statistically sound approach was not used. USEPA's approach for estimating food waste composting is also vague. It states that food waste composting data published by state agencies were used to estimate the tonnage of food waste composted, and the quantity of food waste reported as recovered will vary up or down from year to year depending on data availability (USEPA 2013). Once again, few details regarding which state data were used and how this information was incorporated into the estimates were included.

Consequently, it is unclear exactly how studies were selected for inclusion in the USEPA's food waste estimates, and it cannot be determined if there were any biases involved in the study selection. As Ackerman (1997) has pointed out, many of the reports done by Franklin Associates, Ltd. tend to be devoid of documentation for specific estimates, which makes it difficult to evaluate its work in detail. Use of a more formal analytical approach for quantifying food waste, such as the meta-analytic approach used here, should enable USEPA to address those components of that waste stream that cannot be quantified using the materials flow method.

2. Methodology

2.1 Meta-Analysis Background

Meta-analysis and research synthesis, approaches which employ scientific methodology for data gathering and analysis developed specifically for generalizing results across studies, were used to analyze U.S. waste characterization data. In meta-analysis, standardized effect sizes are used to compare, on the same scale, the results of multiple studies in which a common effect of interest has been measured (Koricheva et al. 2013). An effect size is a statistical index which is comparable across studies, represents the magnitude and direction of the relationship of interest, facilitates calculation of its precision, and is independent of the original scale of measurement used.

After an effect size is calculated for each study, an aggregate (or pooled) effect size across all studies is determined by weighting the precision of each individual effect value so that studies with greater precision are given higher weight than those where effect sizes are estimated with lower precision (Lipsey and Wilson 2001, Koricheva et al. 2013). The main steps of meta-analysis are:

1. Specify question of interest
2. Identify and retrieve eligible studies based on a thorough and unbiased literature review; select studies based on pre-defined inclusion and exclusion criteria
3. Abstract data from eligible studies using a set coding scheme
4. Analyze data in terms of individual effect sizes and aggregate mean effect sizes and determine if outcomes are heterogeneous among studies; perform moderator analyses or meta-regression to examine causes of variation among studies as needed
5. Interpret findings

2.2 Searching for Waste Characterization Sort Studies

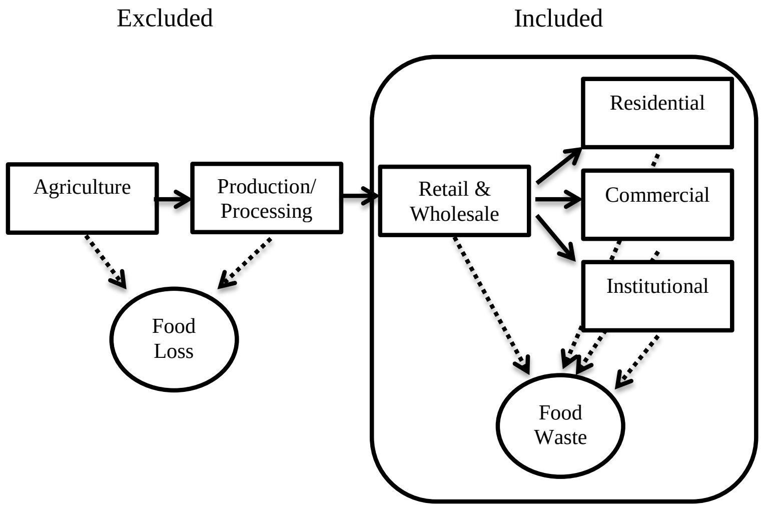
Waste characterization data from state, county, and regional waste characterization studies were found using the Google search engine. Primary search terms were ‘waste sort’, ‘waste characterization study’ and ‘waste composition study’. The search also targeted websites listing waste characterization studies (Appendix A, Table 1). After an initial selection using search terms and study titles, the methodology and results sections were carefully reviewed to ensure studies met selection criteria (see below).

2.3 Inclusion Criteria and Coding

Selection criteria for inclusion were developed prior to choosing or discarding studies. All studies not meeting all selection criteria were excluded and the reason for exclusion was noted (Appendix A, Table 2). Inclusion criteria were: (1) followed general principles and methods outlined by ASTM for waste characterization studies; (2) contained compositional data for food waste based on weight and enabled percentage (by wet weight) of food waste to be determined; (3) performed at a municipal scale (e.g., city, county, state); (4) performed post-recovery of recyclables; (5) involved sampling at the disposal (or transfer) site rather than at the generation point; (6) examined only MSW (residential, institutional, and commercial); (7) involved manual sorting of samples (not visual); (8) confidence intervals and sample sizes were provided; (9) used a standard, comparable definition of food waste; (10) conducted in the U.S; and (11) conducted between 1989 and 2013 in order to capture a 25-year span.

An important selection criterion was that studies focused only on MSW. Some food waste and loss are not included as MSW food waste (Figure 4). Industrial food loss (agriculture, production and processing) is not considered MSW, and it is generally not managed with MSW. The industrial sector faces particular circumstances and regulations, making industrial food waste management different than food waste in MSW. Considerable amounts of industrial food waste are diverted from the waste stream; one estimate was 95 percent of food waste from manufacturers did not go to landfills, with 70 percent used for animal feed (BSR 2013). Agricultural products that are not perfect enough for supermarkets may be sent to shops that are not particular about food appearance, turned into juice or natural sweetener, or fed to animals (Stuart 2009). MSW food waste data do not include wastes that escape through pathways other than MSW systems, such as through home composting, food disposals, or food fed to animals. This approach is consistent with how USEPA quantifies U.S. MSW (USEPA 2013). The present analysis includes avoidable (food that was edible prior to disposal), possibly avoidable (food that some people eat and others do not), and unavoidable (food that is not edible under normal circumstances) food waste (WRAP 2011).

Figure 4. Included and Excluded Food Losses and Waste



Information coded for each study included: study ID number, name, author(s), year, date of publication, scale (county, state, region), state, region (if not at the state level), population of region, sectors included (all MSW, residential, commercial/institutional), residential type (single or multi-family), geographical classification (urban, rural), sampling season, number of samples, average sample weight, and the proportion of food waste as determined from the sampling and its 95% confidence interval. Data on waste shed disposal quantities were recorded as reported in each individual study. These tonnages allow for the determination of the total food waste disposed annually in the waste shed, and for the calculation of pounds of food waste disposed per person per day.

2.4 Per Capita Disposal Rate Calculations

Food waste disposal tonnages and daily disposal rates were determined by Equations 1 and 2, respectively. The food disposal rate represents all food waste disposed in the MSW stream from residential, institutional, and commercial sectors, consistent with the USEPA's estimates of per capita wastes.

$$F = T * P_f \quad \textit{(Equation 1)}$$

F = food waste disposal in a region (in tons)

T = total waste disposal in a region (in tons)

P_f = overall proportion of food waste from study

$$F_r = (2000 * F) / (365 * Po) \quad \text{(Equation 2)}$$

F_r = daily food disposal rate (pounds/person/day)

F = food waste disposal in a region (in tons) (from Eq. 1)

Po = population of region (census or equivalent)

2.5 Meta-Analytic Modeling

2.5.1 Calculating effect sizes

The effect size calculated for each study was a function of the proportion of food waste in the total waste. Meta-analytic approaches for proportions can be grouped according to how they model within-study variability (Trikalinos et al. 2013). The discrete likelihood method, which requires complex computer programming skills and is not commonly used, models the proportion of events in a study using generalized linear mixed models (Wallace et al. 2009). A second approach to modeling within-study variability approximates within-study variability with a normal distribution. Normal approximation introduces bias or has poor statistical properties when the proportion is close to zero or one, or when the study sample sizes are relatively small. Here, the approximation method was used with a variance stabilizing transformation (arcsine transformation); this transformation is a standard means to minimize potential bias associated with the approximation method (Trikalinos et al. 2013). Trikalinos et al. (2013) studied the performance of various meta-analytic approaches for proportions and rates and concluded that the variance stabilizing transformation (arcsine for proportions) should be used for modeling within-study variability for proportions if discrete likelihood methods are unavailable.

The following equations were used to calculate effect size, variance, and the inverse variance weight for proportions (Trikalinos et al. 2013):

$$E_{sn} = \text{arcsine}\,(\sqrt{(P_f)}) \quad \text{(Equation 3)}$$

$$\text{Variance}\, E_{sn} = 1/4n \quad \text{(Equation 4)}$$

$$\text{Inverse Variance Weight} = 4n \quad \text{(Equation 5)}$$

E_{sn} = effect size $_{\text{arcsin}}$

P_f = overall proportion of food waste per study

n = total number of samples per study

The aggregate (pooled) mean effect size across studies was determined by weighting each individual effect size by a term that represents its precision, the inverse variance weight (Equation 5) (Lipsey and Wilson 2001). In this way, more precise studies contribute to aggregate effect size estimates to a greater extent than less precise ones. Variance stabilizing transformations yield summary proportions that have to be back-transformed to the raw proportion scale using the inverse transformation (Trikalinos et al. 2013). So, summary proportions transformed using the arcsine method were later converted back to standard proportions. Back transformation calculations were:

$$P_f = (\sin E_{sn})^2 \quad (Equation\ 6)$$

In addition to using the proportion as an effect size, the per capita food waste disposal rate (F_r) was also aggregated for the sample group of studies surveying all MSW, where possible. These calculations allow comparisons to be made from one waste shed to another, and to rates estimated by USEPA. The sample mean, which was based on a large sample size, was assumed to be approximately normally distributed and sample size was used as a proxy for variance. This was based on the assumption that sampling variances were equal, which is probably not valid because variances are almost never equal across studies. Therefore the meta-analysis outcomes could be biased to an unknown extent (Mengersen and Gurevitch 2014). However, this statistical technique was used as a tool to assess if disposal rates show similar trends as proportions across time and regions, and focus was placed on confidence intervals, rather than point estimates. The following formulas were used to calculate effect size, variance, and the inverse variance weight for the per capita disposal rates:

$$E_{rate} = F_r \quad (Equation\ 7)$$

$$Variance\ E_{rate} = 1/n \quad (Equation\ 8)$$

$$Inverse\ Variance\ Weight = n \quad (Equation\ 9)$$

E_{rate} = effect size $_{rate}$

F_r = daily per capita food disposal rate (pounds/person/day)

n = total number of samples (per study)

2.5.1 Aggregating mean effect sizes

Fixed and random effects models, as described in Hedges and Vevea (1998), are generally used in meta-analysis to calculate the aggregate (pooled) mean effect size across studies. Fixed effects models treat the effect size as fixed, while random models treat the effect size parameters as if they were a random sample from a population of effect parameters. Generally, the homogeneity of effect size parameters influences the model used. If all studies estimate common effect size parameters, a fixed effects model is appropriate; however, if there is heterogeneity among studies, including that from studies not being exactly identical regarding methodology or characteristics of included samples, then the random effects model is appropriate (Viechtbauer 2010). Random effects models are also recommended when analysts intend to draw conclusions that are generalizable beyond the observed studies (Hedges and Vevea 1998). Most meta-analyses now use the random effects model (Trikalinos et al. 2013).

So, a continuous random effects model was used to determine aggregate mean effect sizes. An assessment of overall heterogeneity (variation in study outcomes between studies) was then performed using Cochran's Q, calculated as the weighted sum of squared differences between individual study effects and the pooled effect across studies, with the weights being those used in the pooling method (Hedges and Olkin 1985). When a significant level of overall heterogeneity was found, a meta-regression was performed using a mixed effects model to determine if specific moderators explained any of the heterogeneity. Mixed effects models are random models which allow for the inclusion of moderators to determine if the moderators account for heterogeneity in the effects (Viechtbauer 2010). The specific estimator used in the meta-regression was the restricted maximum likelihood estimator. Tests for the amount of heterogeneity explained in the model by the moderators and for the amount of residual heterogeneity were calculated in the meta-regression, along with tests of each coefficient's individual effect on the proportion (or rate). The calculations were performed using the open-access software program OpenMEE (http://www.cebm.brown.edu/open_mee). A total of four meta-analyses were performed (Table 14).

Table 14. Analyses Performed

Group Name	Group Description	Aggregate Mean Effect Size	Moderators
Total MSW	Studies with samples taken of wastes containing all MSW (residential, commercial, institutional)	Proportion of food waste in the disposal stream	Year, Region
Sector	Studies with samples differentiating between sectors (residential and institutional/commercial)	Proportion of food waste in the disposal stream	Year, Region, Sector
Geographical Classification	Studies with samples differentiating by geographical classification (urban, rural)	Proportion of food waste in the disposal stream	Year, Region, Geographical classification
Per Capita Rate [a]	Studies with samples taken of wastes containing all MSW (residential, institutional, commercial) that enabled rate calculations	Pounds of food waste disposed per person per day	Year, Region

[a] This group is the same as the Total MSW group, with the exception of two studies that lacked sufficient data for per capita rate calculations.

Study samples were grouped based on characteristics of the samples (total MSW, sectors sampled, geographic classification) to ensure effect sizes for each group were independent (no more than one effect size from any subject sample), ensure equitable comparability within a group, facilitate moderator assessment, and allow for valid statistical modeling. Each group was meta-analyzed separately.

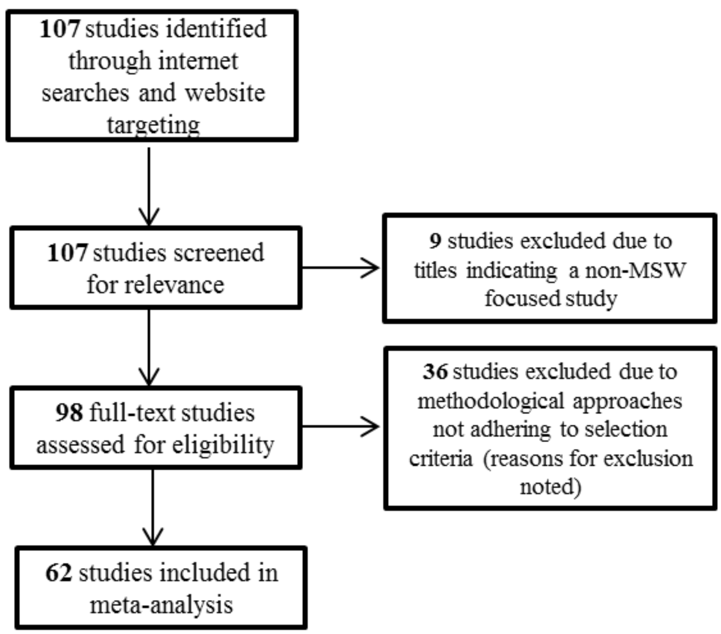
2.6 Comparison to USEPA Estimates

Estimates of food waste disposal from the USEPA's 'MSW in the U.S.: Facts and Figures' quantification reports were collected. Pounds of food waste disposed per person per day were calculated from these data (Equation 2). The USEPA explicitly states that its waste assessments describe wastes from residences, businesses, and institutions (MSW), and the accounting does not included hazardous wastes, dedicated construction and demolition debris (C&D), sewage sludge, and industrial wastes (USEPA 2013). This is consistent with the waste streams analyzed by the waste characterization studies included here for the total MSW and per capita groups, so it is possible to compare the aggregate findings from the waste characterization studies to USEPA estimates.

3. Results

Meta-analyses indicated that food waste proportions have generally been increasing, and the west region generates more food waste (proportionally and as a rate) than the other U.S. regions. One hundred and seven waste characterization studies were examined; 45 were eliminated because they did not meet selection criteria (Figure 5; Appendix A, Table 2) and 62 were included in the meta-analysis (Appendix A, Table 3). Some of the studies had multiple sampling groups in them, so there often was more than one effect size calculated per study (although effect sizes from the same study were not meta-analyzed together). The specific studies which were included in each meta-analysis are described in Appendix A (Appendix Tables 4 through 7). The included studies represent a good coverage of U.S. regions; 20 states were represented. The studies spanned from 1989 to 2013. Most of the included studies were conducted over multiple seasons, so if any seasonal variability existed, it would be accounted for. However, most recent waste sort studies have concluded that seasonal differences in MSW composition are not statistically significant (R.W. Beck 2010, WRAP 2011).

Figure 5. Number of Studies Screened and Assessed for Inclusion



3.1 Meta-Analysis of Total MSW Group

The mean effect size for the proportion of food waste disposed from 1995 to 2013 in the U.S. was 0.147 (+/- 0.010) based on the random effects model (Table 15). Forty-nine studies were included in the total MSW group, conducted from 1995 to 2013. A total of 20,251 samples were included in the meta-analysis, representing about 4,050,200 pounds of sorted waste. There was substantial variation among studies, with the proportion of food waste ranging from 0.071 to 0.228. Significant heterogeneity was found ($Q=144.014$, $p<0.001$).

A significant amount of heterogeneity in the meta-regression model was explained by the moderators (year, region) ($R^2=45.69\%$, $Q_M=19.809$, $p<0.001$). There also was significant residual heterogeneity ($I^2=46.12\%$, $Q_E=77.991$, $p=0.002$), indicating that other moderators may influence food waste proportion. Food waste proportion increased with time, with study year a positive term in the model, indicating a significantly greater proportion of food waste over time ($\beta=0.005$, $z=4.112$, $p<0.001$). The proportion of food waste in the central and western regions were not significantly different than in the east (central: $\beta=-0.009$, $z=-0.484$, $p=0.629$; west: $\beta=0.028$, $z=1.776$, $p=0.076$). Mean effect sizes for each region are given in Table 15.

Table 15. Total MSW Group: Aggregate Mean Effect Sizes

Central Region (n=13)	Aggregate Mean Estimate	0.137
	95% Confidence Interval	0.120, 0.155
West Region (n=17)	Aggregate Mean Estimate	0.153
	95% Confidence Interval	0.140, 0.167
East Region (n=19)	Aggregate Mean Estimate	0.139
	95% Confidence Interval	0.117, 0.163
Overall Aggregate (n=49)	Aggregate Mean Estimate	0.147
	95% Confidence Interval	0.137, 0.157

3.2 Meta-Analysis of Sector Group

Seventy-four studies were included in the next group of studies, those characterized by sector (separate samples from residential and institution/commercial sectors). Studies were conducted from 1989 to 2013. The mean effect size for the proportion of food waste disposed from 1989 to 2013 for this group of studies was 0.181 (+/- 0.016), based on the random effects model (Table 16). Significant heterogeneity among studies was found ($Q=423.298$, $p<0.001$). The total number of samples was 13,962, representing about 2,800,000 pounds of sorted waste.

A significant amount of heterogeneity in the meta-regression model was explained by the moderators (year, region, sector) ($R^2=63.49\%$, $Q_M=77.213$, $p<0.001$). There also was significant residual heterogeneity ($I^2=60.59\%$, $Q_E=181.745$, $p<0.001$), indicating that other moderators may influence food waste proportion. Food waste proportion increased with time, with study year a positive term in the model, indicating a significantly greater proportion of food waste over time ($\beta=0.004$, $z=3.036$, $p=0.002$). The proportion of food waste in the west was significantly different than that in the east ($\beta=0.144$, $z=6.881$, $p<0.001$). The proportion in the central region was not significantly different than in the east (β=0.022, z=0.974, p=0.330). The proportion of food waste was higher for residential samples compared to commercial/institutional sectors, but this difference was not significant (β=0.017, z=1.148, p=0.251). Mean effect sizes for each region and sector are given in Table 16.

Table 16. Sector Group: Aggregate Mean Effect Sizes

Residential Sector (n=36)	Aggregate Mean Estimate	0.182
	95% Confidence Interval	0.160, 0.206
Commercial Sector (n=38)	Aggregate Mean Estimate	0.178
	95% Confidence Interval	0.157, 0.199
Central Region (n=25)	Aggregate Mean Estimate	0.139
	95% Confidence Interval	0.126, 0.153
West Region (n=38)	Aggregate Mean Estimate	0.224
	95% Confidence Interval	0.203, 0.245
East Region (n=11)	Aggregate Mean Estimate	0.123
	95% Confidence Interval	0.111, 0.134
Overall Aggregate (n=74)	Aggregate Mean Estimate	0.181
	95% Confidence Interval	0.165, 0.196

3.3 Meta-Analysis of Geographical Classification Group

Eighteen studies were included in the next group of studies, those characterized by geographical classification (separate samples from urban and rural areas). Studies were conducted from 1999 to 2009. The total number of samples was 2,945, representing about 589,000 pounds of sorted waste. The mean effect size for the proportion of food waste disposed from 1999 to 2009 for this group was 0.153 (+/- 0.018) based on the random effects model (Table 17). The lack of significant heterogeneity (Q=23.146, p=0.145) indicates that there was not a significant difference between studies, including those from rural and urban sectors. Mean effect sizes for each group are given in Table 17.

Table 17. Geographical Classification Group: Aggregate Mean Effect Sizes

Urban (n=11)	Aggregate Mean Estimate	0.155
	95% Confidence Interval	0.131, 0.181
Rural (n=7)	Aggregate Mean Estimate	0.152
	95% Confidence Interval	0.128, 0.178
Overall Aggregate (n=18)	Aggregate Mean Estimate	0.153
	95% Confidence Interval	0.136, 0.171

3.4 Meta-Analysis of Per Capita Food Waste Disposal

Forty-seven studies were included in the per capita group (which included all MSW). Studies were conducted from 1995 to 2013. The total number of samples was 19,902, representing about 3,980,400 pounds of sorted waste. The mean effect size for the food waste disposal rate from 1995 to 2013 was 0.615 (+/- 0.049) pounds per person per day, based on the random effects model (Table 18). Significant heterogeneity was found ($Q=413.319$, $p<0.001$).

A significant amount of heterogeneity in the meta-regression model was explained by the moderators (year, region) ($R^2=38.73\%$, $Q_M= 23.059$, $p<0.001$). There also was significant residual heterogeneity ($I^2=82.6\%$, $Q_E=236.175$, $p<0.001$,), indicating that other moderators may influence the rate. Year led to a greater food waste disposal rate ($\beta=0.005$, $z=1.089$, $p=0.276$), but this was not significant. The per capita food waste disposal rate in the central region was not significantly different than in the east ($\beta=0.072$, $z=1.227$, $p=0.220$). The rate in the west was significantly different than in the east ($\beta=0.233$, $z=4.549$, $p<0.001$). Mean effect sizes for region are given in Table 18.

Table 18. Per Capita Disposal Rates: Aggregate Mean Effect Sizes

Central Region (n=13)	Aggregate Mean Estimate	0.577
	95% Confidence Interval	0.482, 0.671
West Region (n=16)	Aggregate Mean Estimate	0.722
	95% Confidence Interval	0.663, 0.781
East Region (n=18)	Aggregate Mean Estimate	0.503
	95% Confidence Interval	0.436, 0.570
Overall Aggregate (n=47)	Aggregate Mean Estimate	0.615
	95% Confidence Interval	0.565, 0.664

3.5 USEPA Food Waste Disposal Estimates

3.5.1 USEPA estimates

USEPA modifies its materials flow model annually and estimates for previous years are adjusted accordingly, so multiple estimates exist for each year (Figures 6 and 7). Most differences for annual estimates are not large, but changes in estimates for 1995 made in 1997 were strikingly different than those reported in 1996. In the 1996 report, 13,450 thousands of tons of food waste were reported as disposed for the year 1995 (0.28 pounds/person/day); this value was amended to 21,230 thousands of tons of food waste in the 1997 report (0.44 pounds/person/day). The 1996 report estimates 13,200 thousands of tons of food waste were disposed in 1990 (0.29 pounds/person/day), while the 1997 report estimates the same value to be 20,800 thousands of tons (0.46 pounds/person/day). USEPA noted that food waste generation was increased from earlier report versions due to increased population and revised commercial sampling study data.

Most reports (regardless of model year) report the same tonnages of waste disposed for 1960 through 1975. Often, it appears that the same models are used for several years in a row before they are amended. For instance, the 2009 and 2010 models appear to be consistent with each other and report the same tonnages for each year. The same is true for the 1999, 2000, and 2001 models, as well as the 2007 and 2008 models. Appendix A, Table 8 shows the USEPA estimates in detail.

USEPA estimates for the per capita food waste disposal rates and the proportion of food waste in the disposed waste stream have generally increased since 1980. The correlation of the most recent USEPA food waste disposal variables with time was significant (per capita rate: $r=0.72$, $p<0.001$; proportion: $r=0.80$; $p<0.001$).

Figure 6. USEPA Food Waste Estimates- Proportion of Food Waste in Disposed Waste

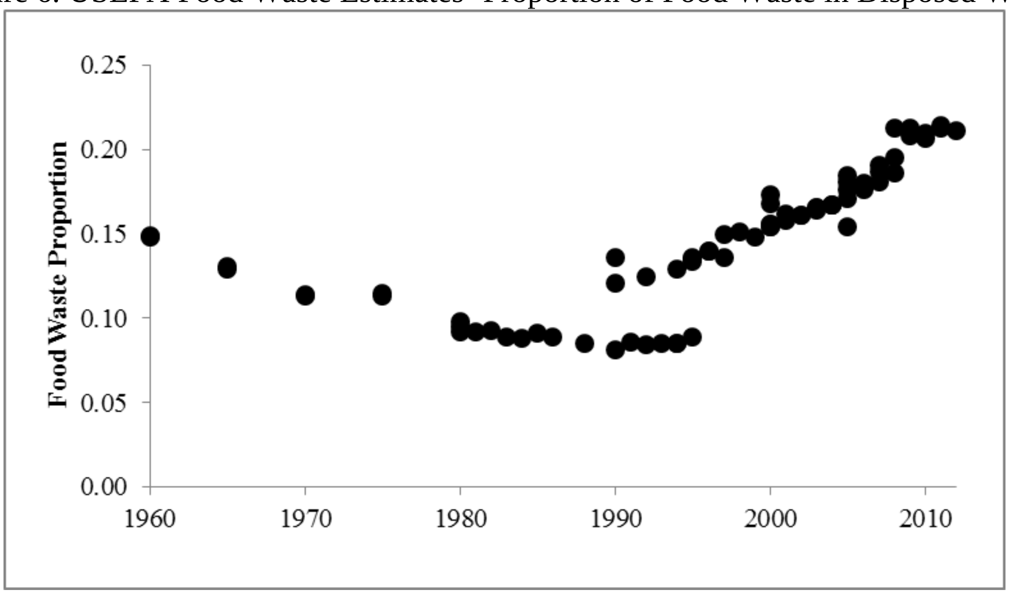


Figure 7. USEPA Food Waste Estimates- Food Waste Disposal Rate

3.5.2 Comparing USEPA estimates to waste characterization study results

The food waste disposal proportions and rates as determined from the waste characterization meta-analyses were compared with the most recent USEPA estimates for the years 1995 to 2013 (Figures 8 and 9); 1995 was selected as the lower year bound because it was the earliest study included in meta-analysis for the total MSW and per capita groups. These groups were used because they are comparable with USEPA data. The annual values for the waste characterization studies were determined by meta-analytically aggregating sort findings from each year (Table 19). There were several years where data were missing (1996, 1997, 2001, and 2003 were missing from the waste sort data and 2013 was missing the USEPA data; data were linearly interpolated in the graphs).

The proportion of food waste disposed in the U.S. as estimated by USEPA is consistently higher than in the waste characterization studies. The aggregate food waste disposal proportion as determined from waste characterization studies was 0.147 for 1995 to 2013. The average proportion based on USEPA estimations for the same period was higher (0.176). The meta-regression showed that the proportion of food waste disposed was increasing significantly with time (Table 19); the aggregate food waste proportion from sort data showed a significant positive correlation with time ($r=0.72$, $p<0.05$). For the same period, USEPA estimates also showed significant positive correlation with time, although the effect was stronger ($r=0.96$, $p<0.005$).

These correlation coefficients were significantly different, based on a Fisher r to z transformation and two-tailed analysis of significance ($z=-.2.59$, $p<0.05$). The aggregate food waste disposal rate as determined from the waste characterization studies was 0.615 pounds per person per day and the average for the same period as determined by USEPA was 0.548.

Table 19. Annual Waste Sort and USEPA Food Waste Estimates

Year	Waste Sort Aggregate		USEPA	
	Proportion [a]	**Per-capita Rate [a]**	**Proportion**	**Per-capita Rate**
1995	0.105 (+/- 0.023)	0.647 (+/- 0.046)	0.134	0.441
1996			0.140	0.439
1997			0.150	0.492
1998	0.144 (+/- 0.105)	0.592 (+/- 0.283)	0.151	0.493
1999	0.147 (+/- 0.031)	0.649 (+/- 0.283)	0.148	0.495
2000	0.119 +/- (0.022)	0.701 (+/- 0.043)	0.173	0.583
2001			0.162	0.505
2002	0.137 (+/- 0.051)	0.583 (+/- 0.165)	0.161	0.518
2003			0.166	0.524
2004	0.132 (+/- 0.027)	0.493 (+/- 0.105)	0.167	0.536
2005	0.136 (+/- 0.058)	0.558 (+/- 0.257)	0.185	0.596
2006	0.139 (+/- 0.080)	0.803 (+/- 0.203)	0.176	0.546
2007			0.191	0.595
2008	0.167 (+/- 0.028)	0.817 (+/- 0.089)	0.213	0.627
2009	0.158 (+/- 0.032)	0.580 (+/- 0.093)	0.213	0.615
2010	0.172 (+/- 0.025)	0.661 (+/- 0.084)	0.210	0.617
2011	0.133 (+/- 0.046)	0.531 (+/- 0.091)	0.214	0.622
2012			0.211	0.616
2013	0.206 (+/- 0.061)	0.526 (+/- 0.147)		
Mean	**0.147** (+/- 0.010)	**0.615** (+/- 0.049)	**0.176**	**0.548**

[a] Aggregate mean as determined by meta-analysis; 95% confidence interval indicated

Figure 8. USEPA and Waste Sorts- Proportion Food Waste in Disposed Stream

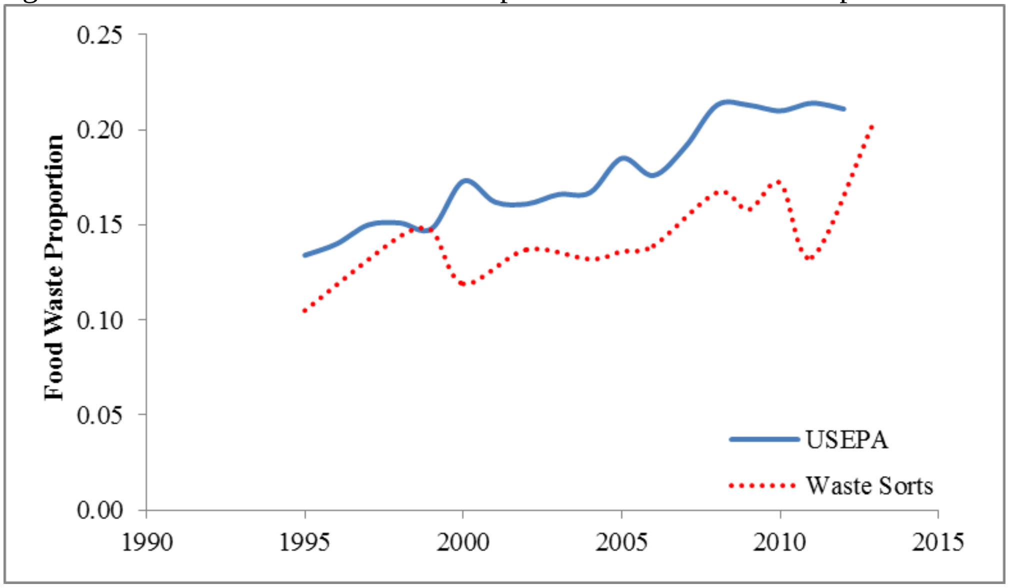
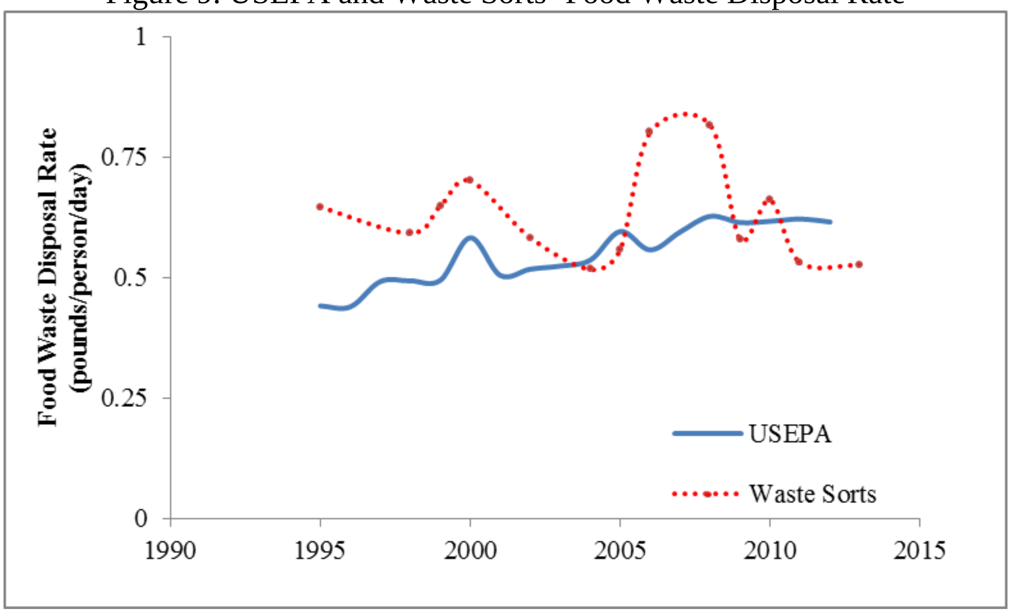


Figure 9. USEPA and Waste Sorts- Food Waste Disposal Rate



Because of the possible introduction of bias into the meta-analytic estimates, particularly for the per capita rate, USEPA estimates were compared to the confidence interval around the point estimates determined by meta-analysis. In five out of 13 years, USEPA estimates for food waste disposal proportion was within the 95 percent confidence bounds for the meta-analysis estimates. USEPA estimates for the per capita disposal rate was within the 95 percent confidence bounds for the meta-analysis estimates for eight out of 13 years. However, the overall USEPA average for the whole period (1995 to 2013) for both proportion and rate was not within the bounds for the aggregate mean as determine from the meta-analysis.

3.6 Total MSW Disposal

A meta-analysis of total MSW (i.e., food waste plus all other residual waste) disposal rates was conducted to provide insight into why specific food waste trends may have been observed. The per capita group was used for these calculations to avoid confusing population trends with disposal trends, and the same meta-analytic methodology as the per capita food waste disposal rate was used. The mean total MSW disposal rate from 1995 to 2013 was 4.249 +/- 0.311 pounds of MSW disposed per person per day. There was a significant amount of overall heterogeneity in MSW ($Q=18757.47$, $p<0.001$). The relationship between MSW disposal rate and year was not significant, although it was negative ($\beta=-0.058$, $z=-1.450$, $p=0.147$), but region was significant, with the west having higher rates of MSW disposal ($\beta=0.857$, $z=2.424$, $p<0.05$).

4. Discussion

4.1 Food Waste in the U.S.

A statistically significant increase in the proportion of food waste in the disposal stream was found, consistent with upward trends in food waste proportions in USEPA estimates. Per capita food waste disposal rates increased with time and total MSW disposal rate decreased with time, but not significantly.

$$P_f = F_d / T_d \quad (Equation\ 11)$$

P_f = proportion of food waste

F_d = food waste disposed (in tons)

T_d = total waste disposed (in tons)

Given Equation 11, if F_d is constant or increasing, and T_d is constant or decreasing, P_f must increase. That is, the increase in food waste proportion is partially related to waste reduction in other components of MSW, which is supported by the downward trend of overall MSW disposal rates. The proportion of food waste is higher relative to these other waste components, even if the amount of food waste disposed remains constant or only slightly increases. Waste reduction of other materials may be due to consumer purchasing choices, material light weighting, longer product durability, and waste avoidance (Tonjes and Swanson 2000). Glass containers are being replaced by plastics, for instance, and many varieties of packaging are thinner. The decline of newspapers due to the availability of online information is likely reducing the amount of paper disposed. Furthermore, over the past 25 years there has been an increase in policies aimed at diverting materials away from disposal, including yard waste disposal bans, bottle bills, more aggressive curbside recycling program, and volume based waste pricing systems (Greene et al. 2010). However, year was not a significant contributor to MSW disposal rates, indicating that these factors may have contributed to decreasing MSW disposal, but the effects were not statistically significant. An extension of the meta-analysis to analyze other materials would provide insight into specific system dynamics, particularly significant increases or decreases in other materials, which may be influencing food waste proportion.

Increases in food waste proportions with time may be partially related to more food being disposed. The per capita food waste disposal rate increased with time, although this effect was not significant. Food waste disposal may be increasing due to more food being allowed to spoil due to it not being used in time, increases in over stocking and over preparation of food (Quested and Johnson 2009), confusion over food labels (Kosa et al. 2007), or misconceptions regarding food safety (Pearson et al. 2013). The significant increase in the proportion of food waste with time is likely due to a combination of less overall MSW disposed, and more food waste disposed, although neither of these trends individually was significant. When trends were examined specifically for six regions where multiple waste sorts were performed over time, these dual trends were observed. Four of the six regions experienced increases in the food waste disposal rate and proportion with time. Total MSW disposal rates decreased in all regions.

There were significant differences between food waste disposal proportions and disposal rates in various regions of the U.S. In particular, all of the analyses examining region as a moderator indicated that the western U.S. has a higher proportion of disposed food waste and disposal rate than eastern and central regions. The high proportion of food waste observed in the west may be partially due improvement in the separation of other materials out of their waste stream, such as traditional recyclables. These robust programs would lead to a large proportion of food waste being left behind in the disposed waste stream, relative to the other materials in MSW. However, the per capita disposal rate of food waste was also significantly higher in the west than in eastern and central regions. This suggests that more food is being thrown away per capita in the west than in the rest of the U.S. It is unclear why this trend is occurring; it is necessary to carefully examine differentiating factors between the west and the other regions to determine which factor is contributing to increased food waste disposal. This finding is surprising as some cities in the western U.S. (e.g. San Francisco, CA) have been acknowledged for their comprehensive recycling and composting programs to divert wastes away from disposal. It is likely that if their recycling programs are exceptionally strong, it will lead to few recyclables being left behind in the disposed waste stream (hence, indicating why a higher proportion of food waste is observed in the west). It also is possible that the western region may generally have more food waste available; so, even if they divert some of this food waste away from disposal, a considerable amount still remains in the disposed waste stream. A waste sort study by Aphale et al. (2015) found that a waste district on Long Island with one of the highest recycling rates in its region also had a higher percentage of recyclables in the disposed waste stream compared to districts with lower recycling rates. Therefore, the high performing recycling district had more recyclables available to them to recycle or dispose; so, even though the residents of that district recovered materials at a higher rate, they also left large amounts of recyclables in their discards. It is possible that a same phenomenon is occurring with food waste in the western U.S.

The proportion of food waste disposed from residential sectors did not differ significantly from that disposed by commercial/institutional sectors. Since MSW disposal tonnages from residential versus commercial/institutional sectors is thought to be between a 60:40 (Bailie et al. 1999) and a 50:50 (R.W. Beck 2000) proportion, considerable tonnages of food waste are disposed by both sectors. This suggests that it may be equally beneficial to target both sectors with food waste recovery or prevention policies. However, there are specific industries that dispose of food waste at much higher proportions than the overall aggregate for all commercial and institutional establishments (Cascadia Consulting Group 2006). Researchers in Massachusetts estimated that highest generators were hotels/lodging (food waste made up 36 percent of disposed waste), elementary and secondary schools (45 percent), fast food restaurants (51 percent), full service restaurants (66 percent), and supermarkets (63 percent) (Draper/Lennon 2002). A waste sort study in Chicago, IL, found that food waste made up 59 percent of disposed waste from restaurants (CDM 2010). In Los Angeles, CA, food waste was found in the highest proportion in restaurant waste (64 percent food waste) and grocery stores (65 percent); educational institutions and hotels/lodging also had high proportions of food waste (45 percent and 33 percent, respectively). Other sectors had very little food waste, particularly businesses (three percent) and retail stores (six percent). Targeting large scale generators may be the easiest way to initiate a food waste management program. Massachusetts has adopted this approach, initially mandating organic waste recovery for commercial and institutional generators producing at least one ton of organic materials per week. The meta-analysis indicated that the aggregate proportion for the sector group was higher than that of the total MSW group. The sector group tends to only include municipally collected wastes, and excludes those dropped off by residents or commercial establishments. Generally, these drop off wastes tend to be higher in C&D debris and lower in food waste, thus explaining why the sector group had a higher aggregate proportion.

There were no significant differences between the proportion of food waste disposed in urban versus rural areas. This finding was somewhat surprising, as urbanization is generally thought to lead to increased food waste generation (Parfitt et al. 2010, Lebersorger and Schneider 2011). It may be possible that it is not urbanization specifically which affects food waste generation, but rather other socio-cultural factors, such as economic development, which may be reflected in the overall standard of living in a country. Iacovidou et al. (2012) point out that economic condition is a critical factor when assessing food waste generation rates; it acts as an indicator of a country's wellbeing and food waste disposal weight has been shown to increase from low to high income countries. Therefore, it is possible that the strong differences between food waste generation in urban and rural populations may be reduced if overall standards of living are high within a country, which is generally the case in the U.S. It would be valuable to extend this analysis to determine if differences between urban and rural food waste disposal differ based on the country of study. Finally, the sample size for the urban-rural dataset were small (n=18). It is possible that strong differences between development type were unable to be ascertained from this sample, so it would be beneficial to continue studying this relationship by adding more samples as more studies are done in the future or by extending the analysis outside of the U.S.

The proportion of food waste in the disposed waste stream as estimated by the USEPA was consistently higher than that determined from the waste characterization studies. It is not clear why these differences exist, particularly because USEPA does not clearly state how they determined their estimates and which studies were utilized for generating estimates. Differences also existed in the food waste disposal rates in terms of pounds of food waste disposed per person per day. However, both the USEPA and waste sort estimates show a general increasing trend over time for the proportion and per capita rates of food waste disposal. Because estimating food and yard waste disposal is not easily accomplished with the materials flow approach, and therefore, must be estimated using alternative means, it would be beneficial for USEPA to follow the meta-analytic procedure used here. This will ensure that their approach is systematic and repeatable, eliminates biases with regards to the inclusion of studies, and provides full clarity with regards to how the estimates were determined. Most importantly, it will help USEPA provide transparency in their model.

Studies indicated that substantial amounts of food waste are disposed annually throughout the U.S.; Appendix A, Table 9 shows the tonnages of food waste disposed per sampling waste shed and per capita food waste disposal values. Because it is not always clear if the sampling region (e.g., county) is representative of a whole region (e.g., state) these data were not extrapolated out to larger regions. The considerable proportion of food waste in the disposed waste stream, and the substantial tonnages that are annually disposed suggest that food waste prevention and diversion away from the disposal stream should be a key priority for sustainable waste systems. If the objective of waste systems is to minimize the amount of materials being disposed in order to ultimately reduce environmental harm, then a focus on food waste should be a key component of this strategy.

4.2 Limitations

Waste composition sort studies rely on sampling because it is neither practically feasible nor desirable to perform waste sorts on all disposed waste. Sampling may lead to random sampling error. There are many factors which may influence whether a municipality performs a waste characterization study, and these factors can influence reported results. For instance, it is likely that those performing studies recognize the need to properly study and manage wastes, implying that they may be more likely to adopt sustainable policies, such as those focused on recycling. So, there could be systematic bias towards higher recovery rates in the waste characterization studies which will affect waste proportions.

There are some areas of uncertainty specific to the waste sorting procedure. During sorting, food waste components are usually separated out of their packaging but there are some items which make separation of food from packaging difficult (e.g., mustard packets, sealed cans). The classification of items which cannot be easily separated from their packaging to the category which proportion by mass prevails is the general practice and that recommended by the ASTM standard (ASTM 2008), but discrepancies may occur when packaging which could have been easily separated is included in the food category or packaging whose proportion is higher than the food inside that is counted in the food category. No estimates are available regarding the dimension of included food packaging within food waste categories from waste characterization studies (Lebersorger and Schneider 2011). Error also may occur through screening, the sieving of waste during sorting. ASTM recommends that sorting be continued until the maximum size of remaining waste particles is approximately 12.7 mm (ASTM, 2008). At this point, apportioning of the remaining particles into corresponding waste components represented in the remaining waste mixture should be done based on a visual estimate of the mass of the fraction of waste components remaining. If this visual apportioning is not done, done poorly, or if sieving is done at various grades, the proportion of food scraps in the waste stream can be underestimated (Dahlen and Lagerkvist 2008), but the exact scale of this error is difficult to quantify.

Waste sorting involves sorting MSW into separate waste categories. However, authorities do not always agree on the definition of MSW and waste categories, and managers may not always count their wastes based on formal definitions (Tonjes and Greene 2012), all of which could lead to inaccuracy in sort results. Most of the waste sorts included in the meta-analysis used fairly consistent MSW and food waste definitions, but there may have been some differences across studies. However, a majority of the studies were done by a few consulting firms that specialize in waste characterization sorts, particularly R.W. Beck and Cascadia Consulting Group. Because the same firms were doing many of the sorts, the methodology and analyses tended to be fairly consistent between studies.

There are some inherent, unavoidable problems with MSW tonnage data, primarily involving the lack of complete data; quantifying this uncertainty is challenging. Some data may be missing due to systematic or intentional errors in waste reports, often from regions relying heavily on private waste collection and facilities (Tonjes and Greene 2012). Some wastes may be disposed outside of the waste shed, thereby preventing their inclusion in the tonnage figures.

Waste reports may also miss data when unlicensed scavengers collect materials or contract carters divert recyclables themselves to enhance revenues.

Pounds per person per day figures are subject to error due to the introduction of population statistics. These population data may not accurately reflect the amount of people living in a municipality at certain times, such as the large summer increases in population observed in some communities that are not reflected in census reports. These summer visitors generate waste which is counted in the municipal data, but they are not included in the population statistics (Greene and Tonjes 2014). It was also assumed that all residents in a waste shed contributed wastes to the sampled waste, which may not always the case (Tonjes and Greene, 2012).

4.3 Future Work

Analyses showed that a considerable amount of food waste is disposed on a regular basis throughout the U.S. These data are important because they indicate how much food waste can potentially be reduced or diverted from the MSW disposal stream. More research is necessary to evaluate the impacts and feasibility of food waste prevention and diversion programs. An LCA and policy discussion are described in chapters four and six. The meta-analyses indicated that despite the explanatory power of some of the variables (year, region), considerable heterogeneity remained, suggesting that food waste disposal may be influenced by other factors, such as education, socio-economic status, or age of residents. Future work to quantify the effects of other variables would be useful.

The technique for quantifying and statistically analyzing the results of waste characterization studies may be expanded to other waste stream components. It is possible to aggregate findings from waste characterization studies to determine the overall disposal proportions and rates for other waste types, as well as to determine if specific moderators are affecting their disposal. It also would be valuable to perform trend analyses on the proportions of other materials in the disposed waste stream and per capita disposal rates to determine how other materials are fluctuating with time. It is necessary to continue performing similar meta-analyses in the future to assess how moderator effects are changing with time, and to determine if food waste proportion continues increasing. Furthermore, as more food waste prevention and recovery programs are initiated in the U.S., it will be possible to use the meta-analysis methodology to assess the effectiveness of these programs, and to determine the differences between regions with food waste collections in place versus those without.

A final area of future work is to quantify the amount of food waste that is disposed outside of the MSW system to provide a more accurate depiction of the total amount of food wasted. Because this study focused on food waste in MSW, wastes that escape through pathways other than the traditional MSW management system (e.g., waste that goes down the drain, food that is composted at home, food fed to animals) were excluded. An Australian study estimated that informal food waste disposal represented 20 percent of Australian food waste flows (Reynolds et al. 2014), which suggests that informal disposal of food waste in the U.S. may be considerable.

5. Conclusion

This was the first study to formally collate and statistically analyze waste characterization studies in a transparent, repeatable, and systematic way using the powerful statistical and conceptual tools of systematic reviewing and meta-analysis. Meta-analysis allows for powerful, unbiased analyses of existing literature (Koricheva et al. 2013) and enables the combination of numerous studies to determine overall trends. This approach serves as a strong alternative to the ambiguous methods used to date to estimate food waste. It is unique in that it focused on food waste disposed in the MSW stream, which makes the findings important for waste management, particularly with regards to planning and policy making. Furthermore, this approach represented a bottom-up approach which integrated smaller scale, real-world sampling studies, as opposed to top-down, large scale, modeling approaches (such as the USEPA's materials flow model) that tend to over-simplify and are rarely validated. It is both essential and urgent that USEPA adopt a similar scientifically transparent and defensible approach to organic waste estimations. The methodology developed here offers considerable advantages over the USEPA approach for quantifying food waste and other waste streams. The methods demonstrated here are more systematic, allow for repeatability and updating, help eliminate biases with regard to the inclusion of studies, and enable full clarity with regard to how the waste estimates are determined. The methodology used here can also be extended to other waste materials to better understand disposal trends for these materials and overall system dynamics.

Food waste was found to make up a considerable proportion of the disposed waste stream, and this proportion has been increasing significantly with time, although the proportion is not as high as that estimated by the USEPA using the materials flow model. The aggregate proportion of food waste disposed in the U.S. from 1995 to 2013 as determined from waste characterization studies was 0.147, which is four-fifths of that estimated by USEPA for the same period (0.179). The per capita disposal rate was not shown to increase significantly with time, although it did have an upward trend with time. There also was no significant effect on food waste proportion by sorts from residential versus institutional/commercial sectors, or from urban compared to rural areas. Region, however, was shown to significantly affect the proportion of food waste in the disposal stream and the per capita disposal rate. The western U.S. had significantly higher proportion and per capita food waste disposal than the eastern and central U.S. When examining total MSW disposal per capita, a general decline with time was observed, but this was not significant.

The study findings indicate that it is necessary to critically evaluate prevention methods and alternative treatments for food waste to determine if environmental, economic, and social benefits can be achieved by diverting it away from disposal and source reducing it. One means to evaluate environmental impacts of alternative treatment technologies is through LCA; an LCA of several food waste diversion technologies is described in chapter four. Furthermore, the high proportion of food waste in the disposed waste stream indicates there is considerable room for improvement with regards to food waste prevention. Prevention may be achieved by various methods, including improved policies and education, both of which are discussed in detail in chapter six. Quantifying wasted food will help bring national attention to the issue, which can greatly advance campaigns to minimize and divert it.

Chapter 4. Environmental Impacts of Food Waste Treatment: A Life Cycle Assessment

1. Introduction

A fundamental challenge of solid waste management is to minimize the potential negative effects of waste systems while maximizing the recovery of useful materials from waste at a reasonable cost (Bailie et al. 1999). Decision makers need sound analyses of the environmental impacts of food waste treatment options to develop successful and effective waste policies (Diggelman and Ham 2003). Here the environmental impacts of waste technologies were evaluated using life cycle assessment (LCA) with the EASETECH modeling software. The goal was to evaluate the environmental impacts of residual waste disposal for a suburban New York (NY) municipality, with a focus on the impacts of adopting separate food waste recovery and treatment. Four food waste treatments were modeled, including waste-to-energy incineration (WTE), two types of composting, and anaerobic digestion (AD), to quantify impacts on climate change, eutrophication, acidification, resource depletion, and stratospheric ozone depletion. This assessment indicated conditions where food waste recovery is beneficial and enabled determination of the management scenario with fewest environmental burdens.

Most previous food waste focused LCAs model the impacts of food waste treatment and do not include residual waste (waste going to disposal) (e.g., Lundie and Peters 2005, Lee et al. 2007, Andersen et al. 2012). Here, food waste diversion from residual waste to alternative treatment was modeled, along with the treatment of the remaining residual waste. An examination of the residual waste stream indicates which materials may be diverted from disposal to enable system improvements. Recycling programs in New York are well-established and mandatory so it is reasonable to assume these efforts will continue. Therefore, examining materials that are still being disposed, such as food waste, are important for systems level improvements, and allow for critical analyses of system dynamics, such as the effects of varying source separation recovery efficiencies. Additionally, inclusion of all residual waste is more realistic than modeling only food waste; this is particularly important for modeling incineration because if only food waste is modeled, net energy production will be quite small due to the high moisture content of these organics (Morris et al. 2013).

No prior peer-reviewed LCA has been conducted for a Long Island municipal waste management system. Most LCA research focused on food waste has been performed outside the U.S. (e.g., Andersen et al. 2012, Bernstad and Jansen 2012b), with most waste LCAs performed in European settings (Laurent et al. 2014b). Food waste is quickly becoming a topic of interest globally, and calls to increase food waste diversion are growing (Levis et al. 2010). Therefore, more research is valuable, especially since there is currently little ongoing research on food waste, especially in the U.S. (Gustavvson et al. 2011).

1.1 Background on Life Cycle Assessment

Systems analysis approaches have been used to evaluate waste management systems, aid in decision making, and assist with policy analysis for several decades (Chang et al. 2011). LCA is a system assessment tool used to identify and quantify the environmental exchanges and impacts of processes in a system. LCA outputs include a set of indicators which simplify and organize inventory results so that they are readily understandable (Owens 1999). The Society for Environmental Toxicology and Chemistry (SETAC) first defined the concept of LCA in 1990, and developed a general methodology for carrying out these studies (Azapagic 1999). The International Organization for Standardization (ISO) defined LCA in their standard 14044 (2006) and described the four basic steps of the assessment:

(1) *Goal and scope definition*, which includes the preliminary assumptions concerning the aim of the study, the functional unit and the boundaries of the system.
(2) *Life Cycle Inventory (LCI)*, which focuses on the quantification of mass and energy fluxes.
(3) *Life Cycle Impact Assessment (LCIA)*, where the environmental impact of the activity is assessed by means of impact indicators.
(4) *Life Cycle Interpretation*, which evaluates possible changes or modifications of the system.

The ISO standard defines LCA methodology in order to increase the transparency and comparability of LCA studies. However, the standard's wording is general and does not give detailed guidance on LCA applications, such as for particular waste components (Bernstad and Jansen 2012a).

Waste LCAs, specifically, quantify the environmental impacts of interconnected waste management technologies from waste generation to final disposal/treatment based on specified waste composition (Chang et al. 2011) and they allow fair comparisons between waste treatment options to be made (Manfredi and Pant 2013). The number of published waste LCA studies and the use of LCA computer models and databases addressing municipal solid waste (MSW) management are increasing rapidly (Cleary 2009, Laurent et al. 2014a). Waste LCA is considered a vital tool for quantifying the environmental benefits and drawbacks of solid waste management technologies (De Feo and Malvano 2009, Wittmaier et al. 2009), and has become one of the principle decision support tools for waste management policy development (Christensen et al. 2007). The European Union (EU) has stated that adopting a life cycle perspective is essential for sustainable waste management (Koneckny and Pennington 2007).

Software tools are essential for LCA; some waste management studies use generic LCA software, such as SimaPro, while others use waste focused software (Laurent et al. 2014b), such as WASTED (Diaz and Warith 2006), WISARD (Buttol et al. 2007), and EASETECH (Clavreul et al. 2013, Levis et al. 2014). A review over 200 waste focused LCAs indicated that about half of them used dedicated waste LCA software rather than generic software (Laurent et al. 2014b).

1.2 Previous Food Waste Life Cycle Assessments

Most waste LCAs include landfilling, recycling, and thermal treatment of MSW, but relatively few model composting and AD (Cleary 2009). There has been limited research on environmental impacts of organic waste management (Levis and Barlaz 2011a), and even fewer studies on food waste in particular (Morris et al. 2014). This may be because LCAs of food waste are particularly complex as they involve both biological and chemical processes and food waste degrades during the waste management processes (Bernstad and Jansen 2012a). There are few LCAs that are performed in the U.S. A review of food waste focused LCAs cited 25 studies and only two were from the U.S. (Bernstad and Jansen 2012a). The authors found considerable differences between study findings regarding optimal food waste management. Table 20 lists recent food waste focused LCAs, their characteristics, and main findings.

Table 20. LCAs Focused on Food Waste

Author Year	Functional Unit [a]	Software [b]	Scenarios	Technologies [c]	Impacts [d]	Country	SA [e]	Main Findings [c, d]
Bernstad and Jansen 2012	1 ton FW	NP	4	AD	AC, ET, EU, GW	Sweden	Yes	AD had impact reductions; FW collection with vacuum bags had greatest EU reductions; collection in paper bags had greatest GW, ET, & AC reductions
Colon et al. 2013	1 Mg FW	Sima pro 7	6	HC	AC, ADP, CED, ET, GW, OD, POF	Spain	Yes	HC is a suitable treatment
Diggelman and Ham 2002	100 kg FW	NP	5	C, WTE, LF, MWW, FWP	AEm, EU, LU, SW, WU, WW, WWa	U.S.	No	C was preferred due to its low H_2O requirements, little wastewater outputs, & capture of nutrients
Grosso et al. 2012	504,000 tons FW & residual waste	Sima pro 7	2	AD, WTE	AC, GW, HT, POF	Italy	Yes	AD had fewer impacts than WTE in most categories
Khoo et al. 2010	570,000 tons FW	NP	4	AD, C, WTE	AC, ET,EU, GW, POF	Singapore	No	AD was preferred, followed by C then WTE
Kim and Kim 2010	1 tonne FW	Total 3.0, Sima pro 7.1	4	WTE, C, AF	GW	South Korea	No	AF & C had low EU & GW impacts
Lee et al. 2007	1 ton FW	IWM-2	5	LF, WTE, C, AF	AC, ET, FE, GW, HT	South Korea	No	LF was main contributor to GW & toxicity
Levis and Barlaz 2011a	1 tonne FW + 550 kg branches	NP	8	C, AD, LF, BLF	EU, GW, Nox, SOx	U.S.	Yes	AD was most preferred option
Lundie and Peters 2005	182 kg FW	NP	5	FWP, HC, C	AC, AE, ET, EU, GW, HT, TE, WU	Australia	No	C had fewest impacts in all categories
Saer et al. 2013	1 tonne of compost (from FW)	Sima pro 7	3	C	AC, CA, ET, GW, NC, OD, SM	U.S.	No	Composting processing stage had highest impacts,

								especially on GW, AC, & ET
Takata et al. 2012	1 ton FW	NP	5	C, AF, AD	GW	Japan	No	AF & AD had low GW impacts
Takata et al. 2013	1 ton FW	NP	6	AD, C	GW	Japan	No	AD had lowest impacts
Vanderm-ersch et al. 2014	1,000 tons FW	NP	2	AD, AF	ALO, FE, FEU, FD, GW, HT, IR, MD, ME, MEu, NLT, OD, PMF, POF, TA, TE, ULO, WD	Belgium	Yes	Treating FW with AD and AF had fewer impacts than AD only
Zhao and Deng 2014	3,584 tons FW	EASE WASTE	3	C, AD, LF	AC, GW, NE, OD, POF	Hong Kong	Yes	LF had high GW impacts; C had high AC & NE impacts

[a] FW = food waste
[b] NP = not provided
[c] AD = Anaerobic digestion; AF = Animal feed; BLF = bioreactor landfill; C =Composting; FWP = food waste processor; HC = home composting; LF = Landfilling; MWW = municipal wastewater;
[d] AC = acidification; ADP = abiotic resource depletion potential; AE = aquatic ecotoxicity; AEm = air emissions; AG = acid gases; ALO = agricultural land occupation; CA = carcinogens; CED = cumulative energy demand; EC = ecotoxicity; ET = eutrophication; EU = energy use; FD = fossil depletion; FE = freshwater ecotoxicity; FEU = freshwater eutrophication; GW = global warming potential/climate change; IR = ionizing radiation; LU = land use; MD = metal depletion; ME = marine ecotoxicity; Meu = marine eutrophication; NC = non-carcinogens; NE = nutrient enrichment; NLT = natural land transformation; NOx = nitric oxide emissions; OD = ozone depletion; PMF = particulate matter formation; POF = photochemical ozone formation; SM = smog; Sox = sulfur oxide emissions; TA = terrestrial acidification; TE = terrestrial ecotoxicity; ULO = urban land occupation; WD = water depletion; WTE = waste to energy; WU = water use; WW = waste water produced; WWa = waterborne wastes
[e] SA = sensitivity analysis

2. Methodology

2.1 Study Goal, Scope and Location Description

The study goal was to evaluate the environmental impacts of residential waste disposal for the Town of Brookhaven (Long Island, NY) to determine if environmental improvement can be achieved by adopting separate food waste recovery and treatment. Brookhaven currently uses WTE as its disposal technology and there is no separation or recovery of food waste; this was considered the baseline scenario, and alternatives to this baseline were evaluated. LCA methodology was used to quantify the impacts of four scenarios in terms of seven impact indicators to determine which food waste treatments provide fewest environmental burdens.

This LCA was performed in accordance with the four phases described in the ISO LCA standard and included a recommended sensitivity analysis (International Standards Organization 2006).

The scale of the LCA was at the municipal level, consistent with most waste LCAs (Cleary 2009, Laurent et al. 2014b). The Town of Brookhaven, which occupies 672 square kilometers, is a suburban municipality with 115,315 households (172 households/square kilometer) (Figure 10). The Town provides residential collection services in 35 districts through contracts with private carters. The waste management program is primarily funded by user fees, collected through Town real estate taxes and landfill tipping fees. Brookhaven owns several waste management facilities, including a transfer station, materials recycling facility, construction and demolition/ash landfill with gas-to-energy recovery, and a small open windrow yard waste composting site. The Town's residential waste is collected curbside, transported to the Town's transfer station to be repacked, and then transported by truck to the Hempstead Resource Recovery Facility for incineration with energy recovery. Incinerator ash is transported back to the Town landfill for disposal (Greene et al. 2013). The Town does not manage wastes generated by commercial establishments or institutions because alternative disposal options are cheaper.

Figure 10. Long Island Map with Waste Management Planning Units

2.2 Functional Unit, Boundaries and Assumptions

The functional unit was one metric tonne (1000 kg.) of MSW disposed by Brookhaven residents and collected curbside with a time frame for emissions of 100 years. This is the typical emission time frame for waste LCAs (Morris et al. 2014). The functional unit included wastes that are currently being disposed and excludes wastes that have been separated for recycling and yard waste composting, and those deposited at drop off locations. These excluded streams are assumed to be identical in all scenarios and are mutually excluding (Grosso et al. 2012).

LCA system boundaries, which represent the interface between the modeled system and the environment, are important to LCA outputs (Morris et al. 2013). System boundaries modeled here include initial residential waste generation, curbside collection, transportation, treatment, and final disposal or recovery. A zero burden LCA approach was used in which all environmental emissions upstream from waste collection were omitted, including those from product manufacture, distribution, and use (Gentil et al. 2010). Therefore, no environmental burdens from material production, agriculture, and consumption were included, consistent with most waste LCAs (Oldfield and Holden 2014). The analysis included direct emissions from waste treatment processes and several indirect emissions associated with the displacement of something in an external system (savings associated with fossil energy substitution by waste-derived energy, the recovery of recyclable materials, and fertilizer substitution). Carbon sequestration in landfills and soils was included. Construction materials and energy for waste facilities were excluded, along with any wastes managed outside the waste system, including food disposed through home composting, waste water treatment, food waste disposal units, or as animal feed. Table 21 summarizes inclusions and exclusions of the study.

Table 21. LCA Boundaries

Included	**Excluded**
-Waste collection and transportation from generation point to final disposal/recovery point (includes fuel combustion) -Waste treatment and/or disposal, including daily operations of treatment facilities -Substitution of fossil fuel energy by waste-derived energy -Substitution of fertilizers by compost -Avoided upstream impacts due to metal recycling (metals collected from WTE ash) -Carbon sequestration in landfills and soils -Energy consumption by waste facilities	-Upstream production and distribution of food and other products -Waste managed outside of municipal MSW system -Construction of treatment facilities and/or machinery -Provision of other external materials (e.g., oils) -Maintenance of equipment -Wastes not collected curbside by the Town -Wastes that have been source separated for recycling or yard waste composting -Commercial or institutional wastes -Waste produced outside of Brookhaven residential waste districts (e.g., village waste)

The marginal unit of electricity used by the waste treatment facilities and the electricity displaced by waste-derived electricity was assumed to come from a mixture of natural gas (81 percent), coal (8 percent), and oil (11 percent). These values were based on the average marginal fuel sources for the northeast U.S. (Siler-Evans et al. 2012). It also was assumed that the collection efficiency of food waste for the scenarios where food waste was source separated and recovered was 70 percent. If Brookhaven was to implement a food waste treatment program, it is likely that food waste would be commingled with the source separated yard waste currently collected for composting. However, because the functional unit excluded all previously source separated materials, the impacts of commingling food and yard wastes were omitted.

2.3 Scenarios

Four food waste treatments (WTE, tunnel and windrow composting, AD) were modeled using EASETECH (Table 22). The specific modeled technological systems represent up-to-date, established technologies that were available in the EASETECH database.

Table 22. Scenarios

Number	Name	Description
1	WTE Disposal	Business as Usual: Current waste management system for Brookhaven. No food waste separation or recovery is performed. All food waste is commingled with residual waste and disposed at a WTE incinerator.
2a	Enclosed Tunnel Composting	Food waste is composted with an enclosed tunnel composting system (all other residual waste is sent to WTE). Compost is produced by aerobic biodegradation. The compost is applied to facilitate plant growth or soil improvement in agricultural contexts.
2b	Enclosed Windrow Composting	Food waste is composted with an enclosed windrow system (all other residual waste is sent to WTE). Compost is produced by aerobic biodegradation. The compost is applied to facilitate plant growth or soil improvement in agricultural contexts.
3	Anaerobic Digestion	Food waste is digested by AD (all other residual waste is sent to WTE). Biogas is produced by hydrolysis, acid fermentation, and methane fermentation. It is used to generate electricity. Digestate is composted aerobically and the final compost is applied to facilitate plant growth or soil improvement in agricultural contexts.

WTE is used widely for MSW on Long Island (Greene et al. 2010). AD and food waste composting, although not widespread in the U.S., are considered potential technologies for food waste because they have been applied broadly and successfully for other wastes, and it is feasible to consider constructing and operating appropriate facilities in the NY metro region. AD plants, especially to treat animal wastes, are becoming more common, partially because biogas is an environmentally desirable fuel (Gomez-Brandon and Podmirseg 2013). Although there are relatively few food waste composting programs in the U.S., composting technologies are well established for yard waste (Platt et al. 2014). Only enclosed composting and AD facilities were included because it is not feasible to use open systems on Long Island due to odor and vector issues. The first scenario analyzed was Business As Usual, which represents current practices and involves sending all waste to WTE, including food waste. Alternative scenarios analyzed potential impacts of implementing source separation of food waste and treatment with alternative technologies, with the remaining waste sent to WTE incineration (Figure 11).

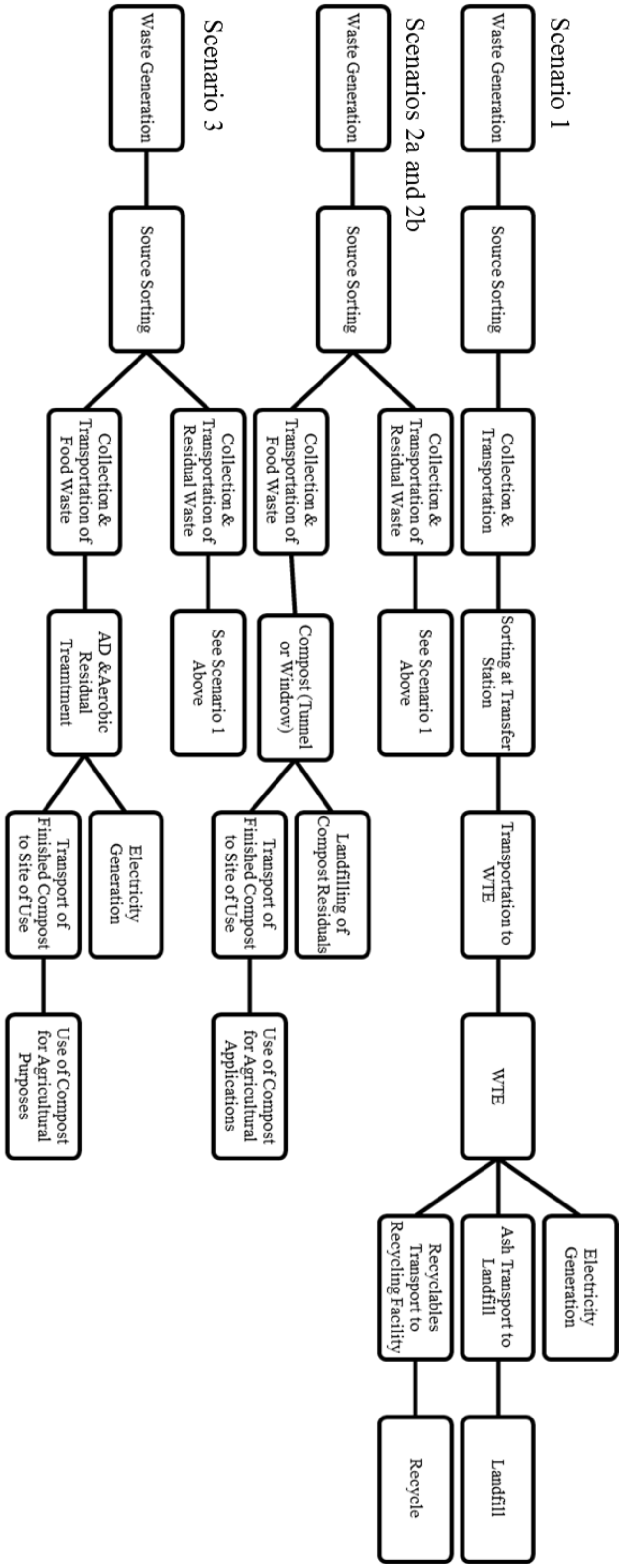
2.3.1 Scenario 1: Business as usual

The first scenario analyzed was Business As Usual where all residual waste is treated with WTE, consistent with current practices. The waste is collected curbside and transported to the Brookhaven transfer station where it is repacked and then transported to the Covanta Hempstead Resource Recovery Facility in Westbury, NY. It is assumed that the average distance from waste collection point to transfer station is 11 km., as this is consistent with Brookhaven's size and location of the transfer station. The distance from the transfer station to the WTE plant is 67 km. The type of incinerator modeled is a grate furnace with wet flue gas cleaning and produced electricity is assumed to offset coal, natural gas, and oil at proportions representing the average marginal profile for the Northeast U.S. electricity grid (Siler-Evans et al. 2012). A fixed electricity recovery value of 23.1 percent is assumed based on values reported by the Hempstead WTE facility. Dioxins and mercury are removed by activated carbon and NO_x is removed by a selective non-catalytic reduction system. Aluminum and iron scrap remaining after incineration is recycled and it is assumed that the distance from the WTE facility to the recycling facility is 81 km. The ash residues from the WTE incineration are transported to the Brookhaven landfill in Yaphank, NY where it is landfilled (LCA is based on a bottom ash landfill described in the EASETECH database). The distance from the WTE plant to the Brookhaven landfill is 67 km.

2.3.2 Scenario 2a: Enclosed tunnel composting

Scenario 2a assumes that food waste is source separated and collected curbside separately from residual waste. Remaining residual waste is collected curbside and treated with WTE, as described in Scenario 1. The separated food waste is transported to the composting site, which is assumed to be located at the Brookhaven waste management facility in Yaphank, NY, where the transfer station is. It is assumed that the average distance from waste collection point to the composting facility is 11 km. The food waste is composted in an enclosed tunnel composting facility where organic materials are mixed and loaded into channels. Tunnel composting systems

Figure 11. Scenario Outline



are aerated containers with forced air through the floor, internal air circulation, and a biofilter. The facility is modelled using an Italian based tunnel composting plant described in the EASETECH database (Boldrin et al. 2009). The bottom of the channel contains a conveyor belt which facilitates the movement of compost from the beginning of the channel to the end and the composting area is kept humid and under pressure to prevent the spreading of odors and bio-aerosols. The feedstock takes about 11 weeks to reach the end of the channel; next, it is unloaded, screened, and laid in open cells for final maturation. Compost rejects are landfilled. The finished compost is used on eastern Long Island for agricultural purposes where it is applied to land using a manure spreader (38 km. from composting facility). The substitution of inorganic fertilizer by this compost and carbon (C) and nitrogen (N) sequestration in soil are included (Bruun et al. 2006, Hansen et al. 2006).

2.3.3 Scenario 2b: Enclosed windrow composting

Scenario 2b assumes that food waste is source separated and collected curbside separately from residual waste. Remaining residual waste is collected curbside and treated with WTE, as described in Scenario 1. The source separated food waste is transported to the composting site, which is assumed to be located at the Brookhaven waste management facility in Yaphank, NY (11 km. from collection point). The food waste is composted in windrows on an enclosed composting pad, which is modelled using a U.S. based windrow composting facility described in the EASETECH database (Komilis and Ham 2004, Andersen et al. 2010, Boldrin et al. 2011). Enclosed windrow composting is essentially a hybrid between outdoor windrow systems and aerated static piles. The facility is equipped with a ventilation system, with off-gases being directed to a biofilter for odor control. The windrows are occasionally turned by a windrow turner. A trommel screen is used to produce fine compost after the curing phase. Compost rejects are landfilled. The finished compost is used on eastern Long Island for agricultural purposes where it is applied to land using a manure spreader (38 km. from composting facility). The substitution of inorganic fertilizer by this compost and C and N sequestration in soil are included (Bruun et al. 2006, Hansen et al. 2006).

2.3.4 Scenario 3: Anaerobic digestion

Scenario 3 assumes that food waste is source separated and collected curbside separately from residual waste. Remaining residual waste is collected curbside and treated with WTE, as described in Scenario 1. The source separated food waste is transported to the AD facility, which is assumed to be located at the Brookhaven waste management facility in Yaphank, NY (11 km. from collection point). The AD facility involves anaerobic in-vessel digestion, followed by aerobic composting of AD residuals. The AD facility is modeled based on a European facility described in EASETECH (Davidsson et al. 2007). The treatment is initiated by mixing the source-separated waste with shredded yard waste structural material. The mixture is placed in the process modules under anaerobic conditions and water is added from the top. Hydrolysis and fermentation processes result in organic acid formation, and materials are then transported to a process tank for methane production by pumping the percolate from the process module. This represents a two-step anaerobic biogas production process, where acid formation and methanogenesis take place in separate compartments. The biogas is burned in an engine at the plant for electricity production to the electricity grid. This recovered electricity is assumed to offset coal, natural gas, and oil at proportions representing the average marginal profile for the Northeast U.S. electricity grid (Siler-Evans et al. 2012). The heat produced is used internally at the plant for heating buildings. After biogas production, process modules are turned aerobic by suction of air through the material initiating a rapid composting process. The exhaust air from the composting process passes through a biofilter. After biogas production and subsequent composting in the closed reactor modules, the compost is placed in open windrows for final maturation. The composted digestate is used on eastern Long Island for agricultural purposes (38 km. from AD facility) where it is applied to land using a manure spreader. Substitution of inorganic fertilizer by this digestate and C and N sequestration in soil are included (Bruun et al. 2006, Hansen et al. 2006).

2.4 Data and Model

The waste composition of the modeled residual waste was based on a 2012 waste characterization study of residual waste performed in the Town of Brookhaven (Aphale et al. 2015). The study performed representative sampling of residual waste (after recycling) from three waste districts in the Town. Aphale et al. (2015) selected these districts because one had a high recycling rate, one had median performance, and one had a low recycling rate. The waste compositions from each district were averaged (simple arithmetic mean) to obtain an overall composition for the Town (Table 23). The specific material categories used were those available in EASETECH and may differ in name from those used in the Brookhaven waste sort, but are assumed to be similar. Although animal and food waste were not distinguished in the

Brookhaven waste sort, it was assumed that animal waste made up one-third of the total food waste, and vegetable-derived waste the remainder (WRAP 2013). Food waste made up less than 14 percent of the total residual waste. Selected characteristics of the materials are provided in Table 23.

Table 23. Modeled Waste Composition and its Characteristics

	% of Waste Stream [a]	**C Fossil**	**C Biogenic**	**N**	**P**	**K**	**Al**	**Fe**	**H_2O**	**Ene-rgy**
Unit [b]	% wet weight	% TS	% TS	% TS	% TS	% TS	% TS	% TS	% wet weight	MJ/kg TS
Aluminum foil and containers (aluminum)	0.98	1.5	13.7	0.4	0.06	0.1	86.1	2.4	18.8	6.8
Clear glass (glass)	2.45	0	0	0	0.006	0.4	0.7	0.5	12.0	0
Other combustibles (other organics/combustibles)	22.99	40.7	13.6	0.9	0.05	0.2	0.4	8.5	9.5	24.4
Wood (wood)	7.63	0.8	51.3	0.8	0.03	0.2	0.4	0.09	15.9	19.0
Hard plastic (rigid plastic)	5.61	79.5	0.4	5.5	0.008	0.02	0.1	0.2	3.3	37.4
Plastic bottles (#1 and #2 plastics)	3.60	76.8	0.4	0.1	0.03	0.04	6.7	0.2	10.5	36.5
Clean cardboard (corrugated cardboard)	7.78	0.2	40.7	0.1	0.01	0.04	1.2	0.3	16.5	15.1
Clean paper (mixed paper)	12.46	0.2	38.1	0.2	0.01	0.07	1.2	0.08	7.4	13.2
Soft plastic (Plastic bags)	6.11	81.6	0.4	0.2	0.2	0.07	0.07	0.03	14.1	40.1
Garden waste (yard waste)	9.23	0.5	17.9	1.2	0.2	1.3	0.8	0	56.0	9.0
Other non-combustibles (other inorganics)	5.61	0.7	0.7	0	0.01	1.0	1.1	0.2	36.6	0
Food cans (ferrous)	2.19	0	0	0	0.02	0.05	21.5	72.7	13.2	0
Vegetable food waste (food waste)	8.89	0.2	47.5	1.9	0.2	1.3	0.1	0.03	77.0	18.3
Animal food waste (food waste)	4.47	1.1	55.4	7.0	1.0	0.5	0.3	0.005	57.1	24.6

Source EASETECH Database

[a] The material names given in parenthesis are those used in the Brookhaven waste sort
[b] TS = total solids

EASETECH, LCA software developed by the Technical University of Denmark, was used. EASETECH has been previously described by Clavreul et al. (2013). The model considers emissions to air, surface water, ground water, soil, and resource consumption with respect to their contribution to defined impact categories. The EASETECH model was chosen because it is a flexible model that allows the modeler to modify modeled processes and adjust default data, allowing for the modeling of specific local facilities. The model includes composting and AD as technological alternatives to WTE and landfilling, important elements for this study. EASETECH also provides strong tools for uncertainty analysis. This assessment was the first study to use EASETECH in a U.S. framework for a full waste management system.

2.5 Inventory and Impact Assessment

An inventory of elementary exchanges associated with the functional unit was determined, and these exchanges were classified and characterized into impact categories. Classification involves assigning life cycle impact results to impact categories (e.g., classifying CO_2 emissions to climate change) and characterization involves multiplying each emission by a factor which quantifies the impact of the emission on the impact category relative to a reference compound (Clavreul et al. 2013). The International Reference Life Cycle Data System (ILCD) approach (2013), the method recommended by EASETECH, was used for impact assessment. This approach utilizes several specific impact assessment methods, as described in the ILCD Handbook (ECJRC 2010). Here seven impact categories were selected for assessment to ensure consideration of multiple types of environmental burdens (Table 24). Cleary (2009) identified global warming potential, acidification potential, eutrophication of surface water, and resource consumption as the impact categories most commonly used in waste LCAs, consistent with those used here. Some LCAs analyze systems using a single environmental indicator, such as global warming potential (Kim and Kim 2010, Takata et al. 2012, Yoshida et al. 2012), but these narrowly focused LCAs may contribute to unidimensional environmental improvement strategies (Ridoutt et al. 2014).

After impact assessment, the results can be normalized by comparing outputs to a given reference, typically a regional value. Here focus was on the relative impacts of each scenario to another, so normalization was not a priority. However, it was performed to determine which impact categories were most affected by each scenario. A review of 222 waste-focused LCAs found that most studies stopped after impact assessment (Laurent et al. 2014b).

Table 24. Environmental Impact Categories Included in LCA

Impact Category	Description [a]	Method [b]	Examples of Classification Data	Charact-erization Factor	Unit	Normal-ization Reference
Climate Change (GW)	Increased absorption of infrared radiation from the earth caused by emissions of greenhouse gases	IPCC 2007	CH_4, CO_2	Global warming potential	kg CO_2 eq.	8,096
Stratospheric Ozone Depletion (ODP)	Destruction of the stratospheric ozone layer by emissions of ozone depleting substances	EDIP	Methane tetrachloro R-10	Ozone depleting potential	kg CFC -11 eq.	$4.14*10^{-2}$
Terrestrial Acidification (TA)	Increased acidity in soil systems by hydrogen ion concentration caused by atmospheric deposition of acidifying substances	AE	NH_3, NO_x, SO_x	Acidific-ation potential	AE	49.6
Terrestrial Eutrophication (TE)	Increased nutrients caused by the deposition of airborne emissions of nitrogen compounds and ammonia	AE	NH_3, NO_x	Eutrophi-cation potential	AE	115
Freshwater Eutrophication (FE)	Increased nutrients caused primarily by the deposition waterborne emissions of nitrogen and phosphorous-containing compounds. P is generally the limiting nutrient in freshwater systems. Large lakes may also be affected by airborne emissions	ReCiPe midpoint	P, PO_4^{3-}	Eutrophi-cation potential	kg P eq.	0.62
Marine Eutrophication (ME)	Increased nutrients caused by the deposition of airborne and waterborne emissions of nitrogen and phosphorous-containing compounds. N is generally the limiting nutrient in marine systems	ReCiPe midpoint	NH_3, N, NO_x	Eutrophi-cation potential	kg N eq.	9.38
Depletion of Fossil Resources (ARF)	Decrease in availability of the total reserve of potential functions of resources (minerals and fossil fuels) beyond their rate of replacement	CML 2012	Coal, Oil, Diesel	Resource depletion potential	MJ	$6.24*10^4$

[a] European Commission 2010

[b] AE = accumulated exceedance

2.6 Sensitivity Analysis

Sensitivity analysis involves evaluating the sensitivity of model results to variations in input data and modelling choices (Clavreul et al. 2012). In the present study input parameters were varied across a range of possible values and the changes in the model outputs were noted (Table 25). The first parameter tested was the sorting efficiency of food waste (the percentage of food waste diverted divided by amount of food waste generated). The next parameters tested focused on transport distances to determine the benefit of siting treatment facilities close to generation or compost use points. Last, the impacts of the marginal energy profile were explored. A profile consistent with that in the mid-Atlantic U.S. was modeled as an alternative to the default northeast U.S. profile.

Table 25. Sensitivity Analyses

Parameter Changed	Default Value	Sensitivity Values
Sorting Efficiency	70%	30%, 50%, 90%
Transport distance from generator to management facility	11 km.	50 km., 200 km., 400 km.
Transport distance from management facility to compost use site	38 km.	100 km., 200 km., 400 km.
Marginal energy profile	Northeast U.S.: 8% hard coal, 81% natural gas, 11% fuel oil	Mid-Atlantic U.S.: 70% hard coal, 29% natural gas, 1% other

3. Results

3.1 Total Environmental Impacts

Attention was focused on measuring the overall impact of each scenario across impact categories and on the contribution each process has to the complete impact profile, particularly the food waste treatment processes. In all scenarios, the climate change, terrestrial eutrophication, and marine eutrophication impacts were positive, indicating net impacts, while ozone depletion, freshwater eutrophication, terrestrial acidification, and resource depletion indicated avoided impacts (Table 26). Net savings were observed for these categories because of the inclusion of indirect impacts resulting from the substitution of materials outside the waste management system (e.g., electricity, fertilizers). Because the whole residual waste stream was modeled, much of the waste within the system was treated similarly in different scenarios

(WTE), as the variation only covered the food waste making up less than 14 percent of the modeled waste (and only 70 percent of this food waste was diverted). However, it is important to model the full waste flow as changes to food waste management will also impact the other residuals management (a comparison of food waste alone is described in section 3.4). So, the relative difference between the environmental performance from one scenario to another are small, but it represents the potential changes that can be achieved when diverting food waste away from WTE.

Table 26. Potential Environmental Impacts (treatment of one tonne residual waste)

Scenario [a, b c]	**GW**	**ODP**	**TA**	**TE**	**FE**	**ME**	**ARF**
Unit:	kg CO_2 eq.	kg CFC-11eq.	AE	AE	kg P eq.	kg N eq.	MJ
1	185	-0.0000026	-0.61	2.40	-0.000035	0.22	-911
2a	204	-0.0000026	-0.62	2.23	-0.0072	0.29	-899
2b	206	-0.0000026	-0.61	2.23	-0.0072	0.32	-885
3	185	-0.0000026	-0.67	2.09	-0.0075	0.28	-949

[a] A negative value indicates impact saving/emission reduction
[b] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources
[c] Scenario 1 = WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

3.2 Scenario Rankings

Scenarios were ranked to identify the most environmentally sound scenario in each impact category (a rank of one indicates best environmental performance). The rankings for each system were then averaged to give an overall ranking across all impact categories (Table 27), similar to the approach used by Diggelman and Ham (2003). Because a study goal was to evaluate if system changes lead to better environmental performance relative to the business as usual scenario, the absolute impact values are of lesser importance than the relative impacts of one scenario to another. This approach is better for system planning, as decision making based on the relative performance of alternative policy scenarios under a range of scenarios is preferred rather than on a single modeled scenario with absolute outputs (Plevin et al. 2014).

Table 27. Environmental Impact Rankings

Scenario [a, b]	GW	ODP	TA	TE	FE	ME	ARF	Average Ranking	Overall Rank
1	1	1	3	3	3	1	2	**2.0**	**2**
2a	2	1	2	2	2	3	3	**2.1**	**3**
2b	3	1	3	2	2	4	4	**2.7**	**4**
3	1	1	1	1	1	2	1	**1.1**	**1**

[a] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources
[b] Scenario 1= WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

There were differences between rankings based on impact category. Scenario 3 performed the best (or tied for the best) in all impact categories except marine eutrophication. On average, based on all impact categories, scenario 1 (WTE) performed next best, followed closely by scenario 2a (tunnel composting); scenario 2b (windrow composting) showed the worst performance. The business as usual scenario performed better than at least one of the alternative scenarios in three impact categories (climate change, marine eutrophication, depletion of fossil resources), indicating that switching to an alternative food waste treatment may increase environmental burdens in three out of seven impact categories; environmental burdens would decrease (or remain unchanged) for the remaining five impact categories. The windrow composting scenario yielded the highest environmental burdens and did not perform better than the business as usual scenario when considering all impact categories together.

3.3 Process-Specific Impacts

Figures 12 to 18 show the contribution of each process in the waste system for each impact category. Scenarios were broken into seven process groups (Table 28). Most impacts are discussed here, although some categories had very low emissions. The importance of each impact is discussed in the normalization section 3.5.

Table 28. Process Groups

Group	Included Processes
Collection	Curbside collection; transport to initial management point (transfer station or alternative treatment facility)
Transport	Transport from transfer station to WTE; transport of treatment outputs to site of use/disposal/recovery; repacking of waste at transfer station
WTE	Waste to energy incineration operations
Alternative Food Waste Treatment	Composting and anaerobic digestion operations
Recycle	Recycling of metals recovered from incineration
Landfill	Landfilling of WTE ash and compost rejects
Compost Use	Application of compost for agriculture and substitution of fertilizers

Figure 12. Climate Change (GW) - Process Specific Impacts

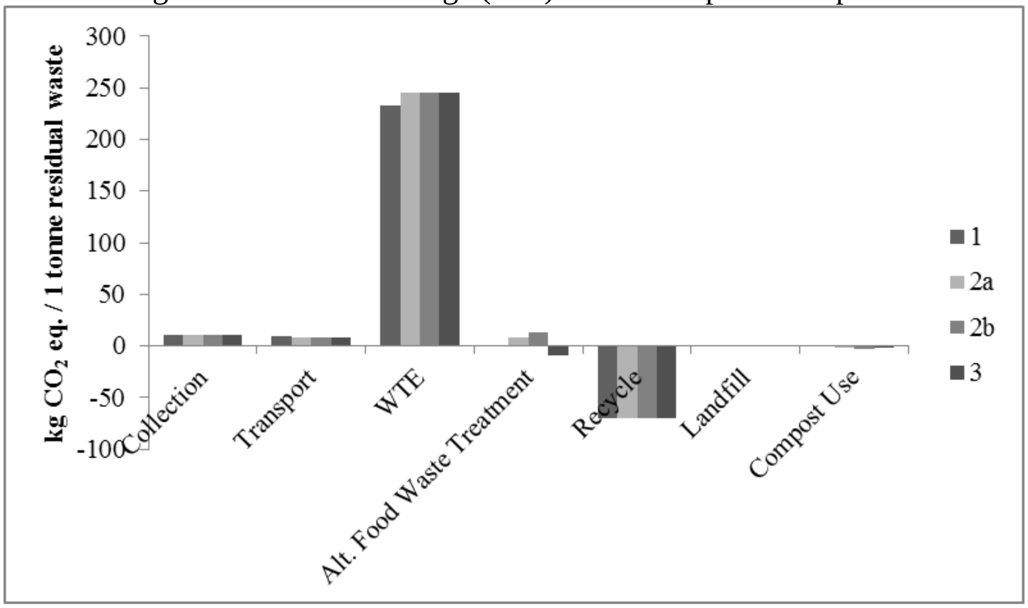


Figure 13. Stratospheric Ozone Depletion (ODP) - Process Specific Impacts

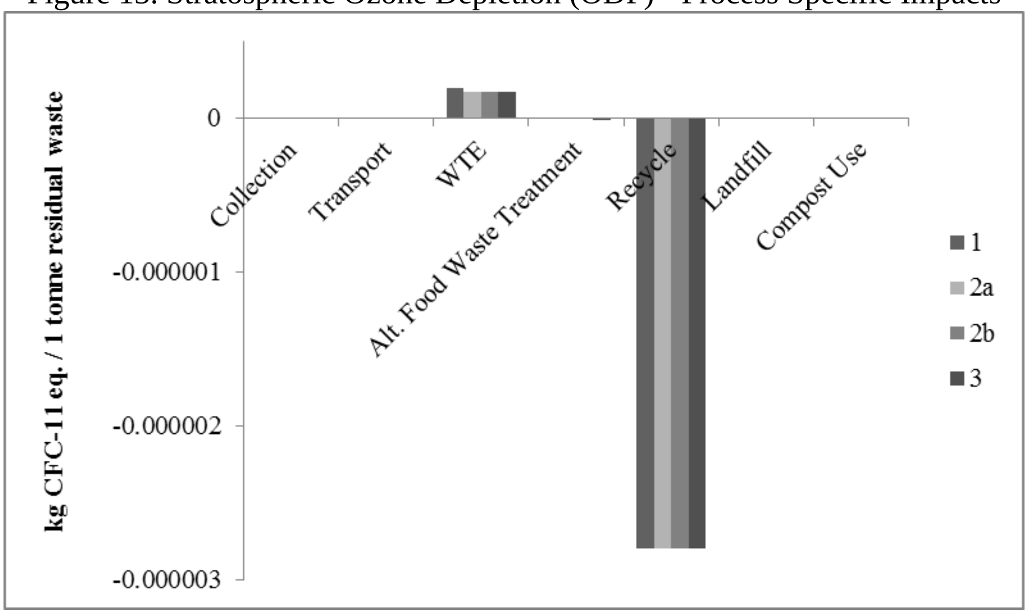


Figure 14. Terrestrial Acidification (TA) - Process Specific Impacts

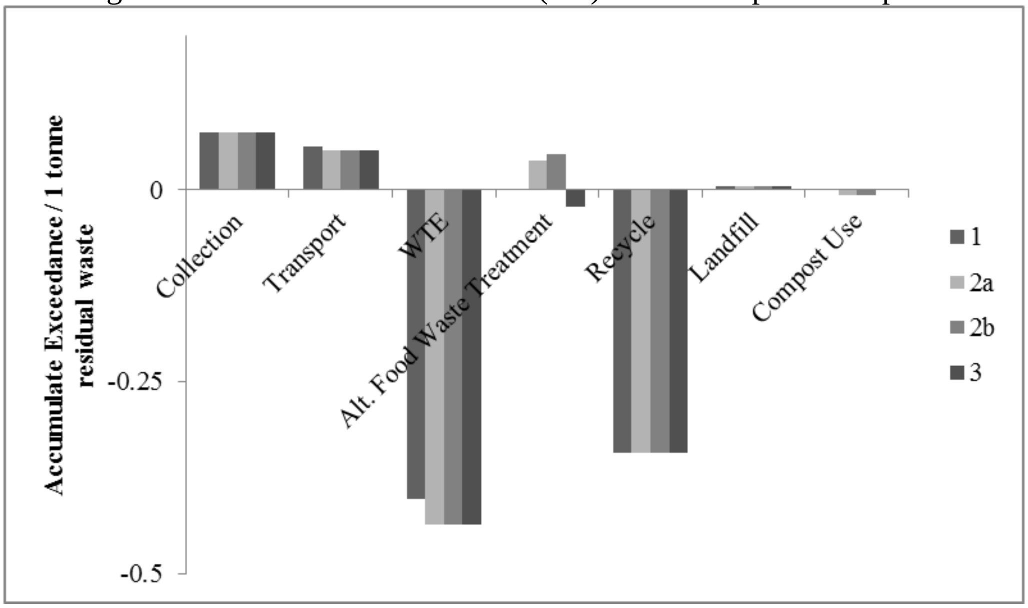


Figure 15. Terrestrial Eutrophication (TE) - Process Specific Impacts

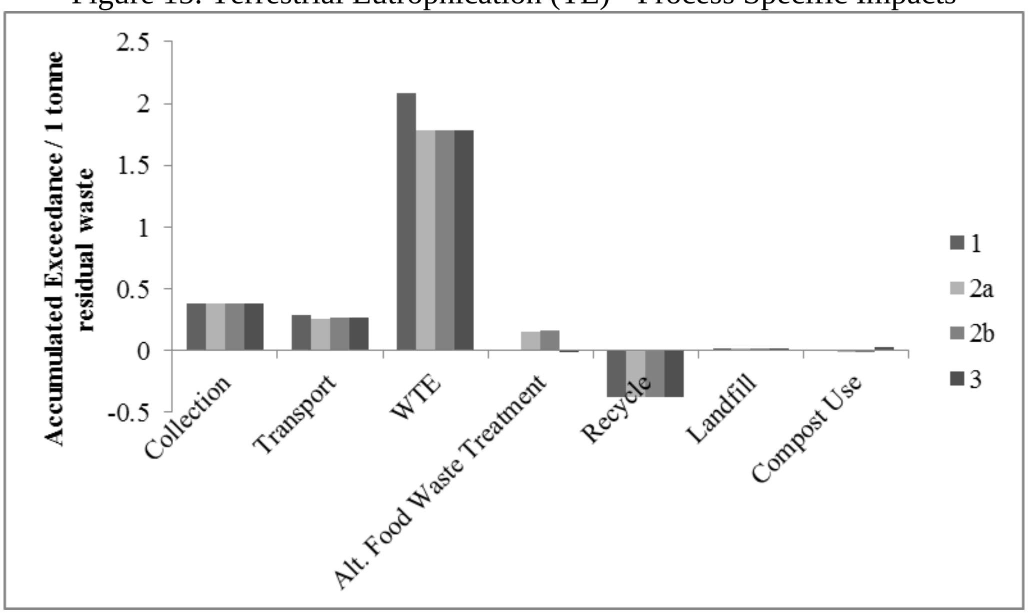


Figure 16. Freshwater Eutrophication (FE) - Process Specific Impacts

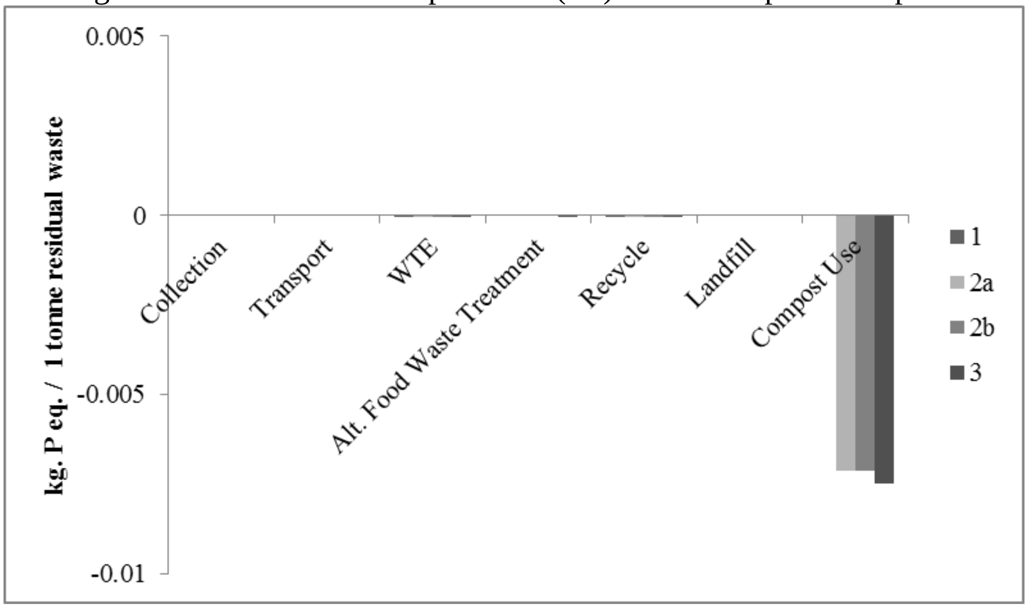


Figure 17. Marine Eutrophication (ME) - Process Specific Impacts

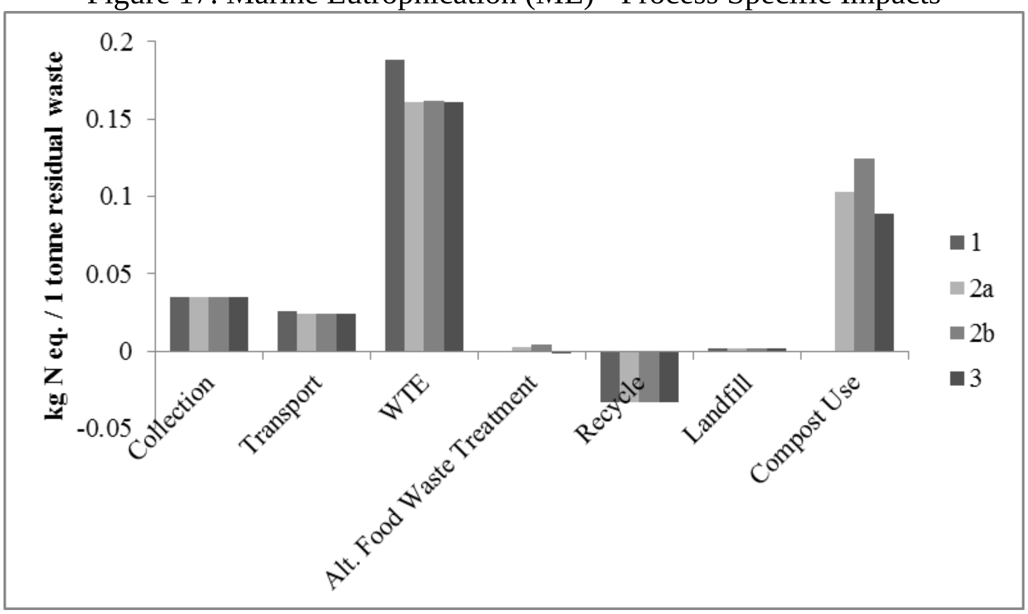
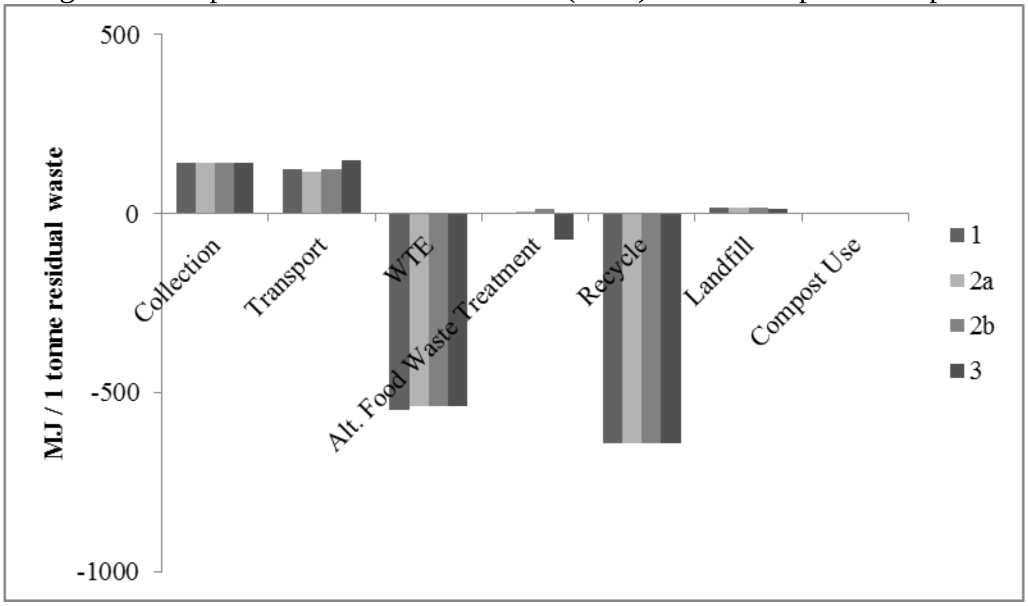


Figure 18. Depletion of Fossil Resources (ARF) - Process Specific Impacts



The process with the greatest impacts differ based on the impact category. Collection and transport contributed relatively moderately to the life cycle impacts in all impact categories. Greatest impacts for these processes relative to the others occured in the depletion of fossil resources, terrestrial acidification, terrestrial eutrophication, and marine eutrophicaiton categories. The fuel consumption during collection and transportation yield NO_x and SO_x emissions. These emissions contributed to the terrestratial eutrophication and acidfication, and marine eutrophication outputs. This fuel use also contributed to depletion of fossil resources and climate change emisisons.

WTE achieved considerable impact savings as well as outputs, depending on the impact category. Emisisons were observed in the climate change category due to stack emissions, primarily of CO_2. Some of these emissions were offset by the substitution of fossil fuels with waste-derived energy, but not enough to yield overall net savings. These emissions can be offset to a greater degree if the emissions from the energy being displaced by WTE have very high climate change impacts or if the efficiency of the WTE facility can be improved. Emissions also were observed for marine and terrestrial eutrophication primarily due to NO_x emissions, although these were offset to a small degree by the replacement of fossil energy. Considerable fossil resource savings resulted from the offset of coal and oil usage. WTE yielded net terrestrial acidification savings due to SO_2 and NO_x offsets from the replaced fossil fuel energy. Savings also were observed for freshwater eutrophication due to savings in phosphate emissions.

The recovery of scrap aluminum and steel from WTE ash provided substantial savings for climate change, primarily due to CO_2 savings. Minimal stratespheric ozone depletion were observed due to CFC-11 savings, but these impacts were small and carry little importance. Savings in the ozone depletion and terrestrial acidification were also observed. Recycling enables the production of materials (aluminum and steel) from raw resources to be offset, leading to the observed savings.

Landfilling had minimal impacts in all categories because the relatively small quantity of materials landfilled. Also, because organic carbon in waste was destroyed by WTE, there were no greenhouse gas emissions from the disposal of the ash in the landfill (Papageorgiou et al. 2009). Although effects were minimal, net impacts rather than savings were observed in all impact categories for landfilling processes.

The three scenarios for food waste treatment and compost use showed the smallest impacts relative to other processes because the proportion of waste treated by these technologies was considerably less than that treated as WTE (906.7 kg. of waste to WTE; 93.3 kg. of food waste to alternative treatment). However, food waste treatment and compost use contributed most to the variation in impacts between scenarios, so differences across the scenarios were important.

3.4 Food Waste Treatment Impacts

Table 29 gives impacts for the alternative treatment of source separated food waste (93.3 kg. of food waste resulting from source separation of 70% of the total food waste in the 1,000 kg. total MSW). The impacts of treating this waste with WTE were also provided for comparison to the alternative treatments. This table only indicates results from the waste treatment process (WTE, AD, composting), not other system components (e.g., transport). Differences exist for climate change impacts. Composting operations yielded net climate change impacts, rather than savings, because they require energy inputs and do not have an electric power output. Similarly, Khoo et al. (2010) found that aerobic composting had higher climate change impacts than WTE due to the high energy consumption composting requires. Morris et al. (2014) found that composting had higher energy impacts than AD for the same reason. For composting, the greatest contributors to climate change were N_2O and CO_2 emissions, partially from energy consumption. The emissions of C and N compounds from indoor composting were reduced compared to outdoor composting due to the use of a biofilter, and the same filter efficiencies were assumed for both composting scenarios. Emissions of SO_2, NO_x, and NH_3 from daily operations (e.g., electricity requirements of facilities) and fugitive emissions which escape through the biofilter contributed to the terrestrial acidification, terrestrial eutrophication, and marine eutrophication impact. Depletion of fossil resources occurred due to the electricity requirements and mechanical equipment in the composting facilities. The differences between the two composting technologies were due to differing electricity requirements (electricity consumption for 2a and 2b are set to be 53.4 kWh/Mg and 143.8 kWh/Mg, respectively).

Table 29. Food Waste Treatment Process Impacts

Impact [a, b]	**Unit** [a, b]	**1**	**2a**	**2b**	**3**
GW	kg CO_2 eq.	-12.54	8.59	12.91	-9.25
ODP	kg CFC-11 eq.	2.10E-8	4.63E-10	8.25E-10	-6.32E-09
TA	AE	0.03	0.04	0.05	-0.023
TE	AE	0.30	0.16	0.16	-0.014
FE	kg P eq.	-1.16E-6	3.02E-07	7.33E-07	-1.64E-06
ME	kg N eq.	0.03	0.0025	0.0039	-0.0013
ARF	MJ	-9.21	7.09	13.31	-73.88

[a] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources

[b] Scenario 1= WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

AD provided net savings in all impact categories due to the replacement of fossil fuel energy by AD-generated energy. The greatest emissions savings for the global warming category resulted from CO_2 savings. Although environmental emissions from AD were reduced due to a biofilter, some fugitive emissions and emissions from AD facility operations occurred. However, direct emissions of NOx, NH_3, and SO_2 emissions from AD were offset by the offset of fossil fuel production. This offset also yielded considerable savings in the depletion of fossil resources category.

Table 30 provides impacts from the use of compost. Compost use refers to land application, fertilizer substitution, and soil C and N sequestration for compost and AD residuals that are composted. Compost use yielded savings in all impact categories except ozone depletion, marine eutrophication, and depletion of fossil resources. These net impacts were partially due to the application of compost to land using a diesel manure spreader. The climate change savings were primarily due to carbon sequestration in soils from compost. The other savings result from substituting compost for commercial fertilizers. On average, the composting scenarios rank better than AD when considering the effects of compost use in all impact categories due to the increased quantity and quality of compost directly from composting processes rather than from composting AD residuals. Because AD utilizes organic material during the digestion phase for energy production, there is less residual material remaining to be composted and land applied than in the composting scenarios; AD residuals also tend to be of poorer nutrient quality (Andersen et al. 2012).

Table 30. Compost Use Process Impacts

Impact [a, b]	**Unit** [a, b]	**2a**	**2b**	**3**
GW	kg CO_2 eq.	-0.78	-2.93	-2.16
ODP	kg CFC-11 eq.	2.07E-11	4.51E-11	1.26E-10
TA	AE	-0.0065	-0.0066	0.00055
TE	AE	-0.0046	-0.0043	0.032
FE	kg P eq.	-0.0071	-0.0071	-0.0075
ME	kg N eq.	0.10	0.12	0.089
ARF	MJ	0.29	0.63	1.74

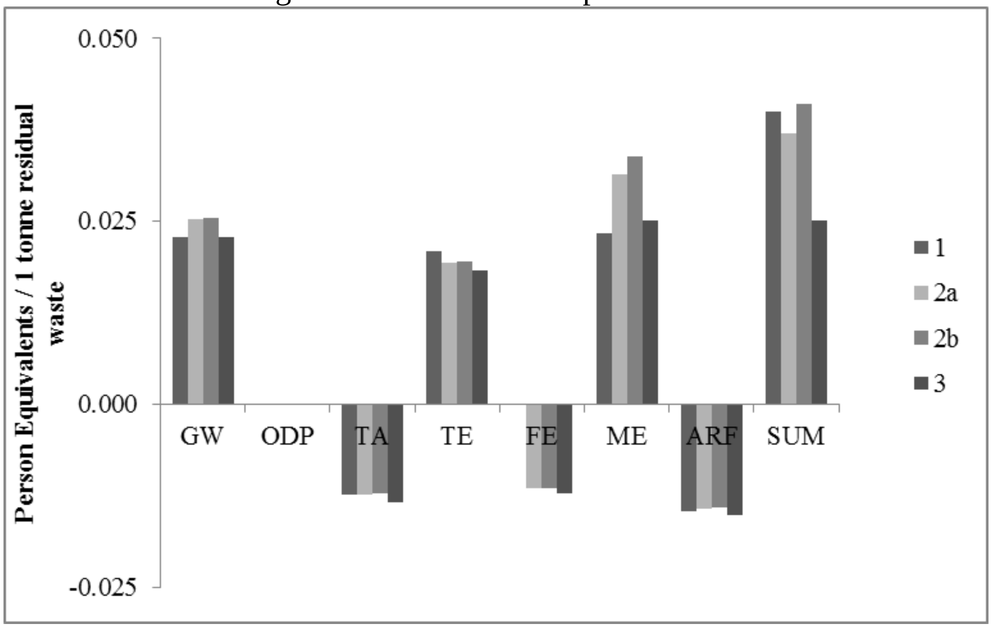
[a] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources
[b] Scenario 1= WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

Composting offers additional benefits that are difficult to quantify through LCA, including weed suppression, increased soil productivity, and water conservation. The LCA literature does not currently have an impact category directly assessing soil quality and productivity, although soil carbon sequestration and synthetic fertilizer displacement are typically included (as they were here) (Morris et al. 2014). It is necessary to qualitatively recognize the additional benefits of compost to soils when examining composting as a technology option, and future efforts to formally quantify them should be made. It is likely that this will improve the performance of composting relative to other technologies.

3.5 Normalized Environmental Impacts of Each Scenario

Normalization to person equivalents allows for comparisons across impact categories. EASTETECH's default normalization approach was used because it was developed specifically for the ILCD 2013 impact assessment method used here (normalization values are provided in Table 24) (Blok et al. 2013). The normalization factors were derived from global and European emission references, so the normalized values may not accurately reflect the Brookhaven per capita impacts. However, using the EASETECH values enable an understanding of the relative impacts from one impact category to another although the absolute values may not fit Brookhaven (Figure 19).

Figure 19. Normalized Impact Profiles



The impact category with the highest normalized effects in all scenarios was marine eutrophication, followed by global warming, and terrestrial eutrophication. These categories also showed the greatest differences across scenarios. The smallest differences across scenarios occurred for ODP. Fossil resource depletion showed the highest normalized impact reductions. Overall, all scenarios showed greater environmental burdens than savings, as indicated in by the sum of normalized impacts across all impact categories for each scenario. The sum shows total impacts if all impact categories are assumed to be of the same importance. Ranking to weight the relative impact of impact categories was not conducted, but could be carried out in the future.

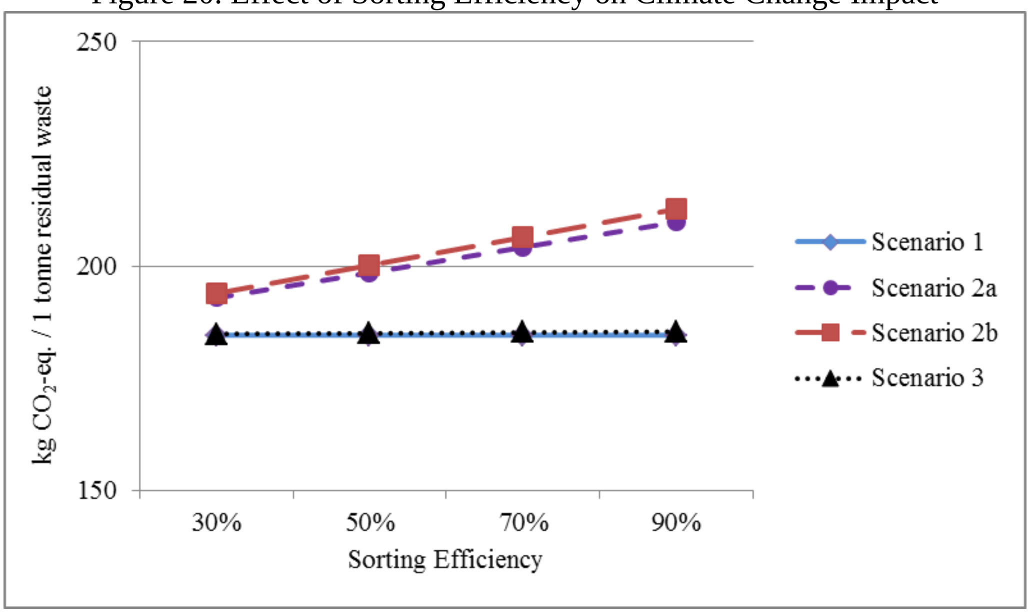
3.6 Sensitivity Analysis

The effects of altering several input parameters on climate change were examined. This impact category was a focus because it is of particular interest in the waste management field (Vergara et al. 2011). It also had one of the highest normalized impacts.

3.6.1 Sorting efficiency

Soring efficiency represents one of the largest sources of uncertainty for this analysis. Using LCA, Yoshida et al. (2012) found that capture rate considerably affected greenhouse gas emissions when examining several organic waste treatment options. Waste sorts have indicated that even in areas with robust recycling programs, considerable amounts of targeted recyclables are still found in disposal streams; a study in Brookhaven found that up to one-third of discarded residual waste was recyclable material (Aphale et al. 2015). Therefore, not only is it likely that a new food waste source separation program will not achieve 100 percent sorting efficiency, but efficiencies may not meet the 70 percent used in the default models. In all scenarios the climate impact increased with increased sorting efficiency (Figure 20), although these increases were not substantial and did not change the rank ordering of the scenarios for the climate change impact category. Because a linear modeling approach was used, most changes in impact results were also linear. Due to capacity sizes for some equipment, it is possible for changes to be non-linear for some technologies, such as collection. Because the baseline scenario has less climate change impact potential than the composting scenarios, as more food waste is diverted away from WTE through alternative technologies, the climate change impact potential of the alternative scenarios increases. WTE has the same climate change impact potential as AD; as sorting efficiency increases, the climate change impacts of AD increase minimally.

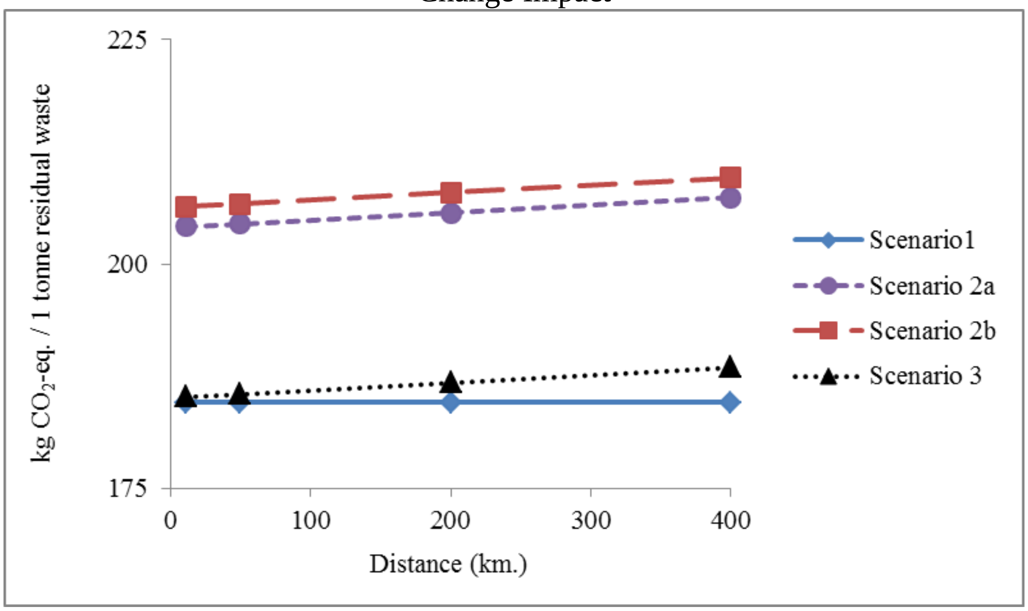
Figure 20. Effect of Sorting Efficiency on Climate Change Impact



3.6.2 Transportation distance

Transportation and collection are the most commonly tested parameters for sensitivity assessments in waste LCAs, although several studies have shown that impacts of waste transport rarely has a large influence on overall system environmental impacts (Grosso et al. 2012, Laurent et al. 2014b), although Lundie and Peters (2005) found otherwise. Distance is an important issue for food waste policy; several recently enacted New England food waste regulations have food waste diversion requirements based on generator distance from an appropriate facility. When examining the impact of increasing the transport distance from generation point to food waste management facility, the climate change impact increased with increased distance, but not substantially (Figure 21). The rank ordering of scenarios for the climate change category did not change. Similar findings were observed when increasing the distance from management facility to the compost use site.

Figure 21. Effect of Transport Distance (from Generator to Management Facility) on Climate Change Impact



3.6.3 Substituted marginal energy profile

Waste-derived energy can substitute for energy from other sources. This substitution may have a large result on LCA results, especially regarding climate change impacts (Bernstad and Jansen 2012a). Change in the marginal energy from one consistent with the northeast U.S. to a profile consistent with the mid-Atlantic U.S. considerably affected climate change impacts (Table 31). By changing the marginal profile from one primarily dependent on energy from natural gas, a relatively clean fossil fuel, to one primarily dependent on hard coal, a high polluting fuel, each overall scenario impact decreased substantially in the climate change category. Considerable savings were observed primarily due to savings obtained during WTE from fuel substitution. The inclusion of waste-derived fuels into the electricity grid offsets production by coal, a highly polluting energy source. Savings for the mid-Atlantic energy profile are greater than those with the Northeast profile, and the rank ordering of scenarios changed with the mid-Atlantic profile. With the northeast energy profile, scenario 1 and 3 had the lowest climate change impacts, followed by 2a, then 2b. With the mid-Atlantic profile, scenario 1 achieved greater impact savings than scenario 3. Scenario 2a still had lower impacts than 2b.

Table 31. Marginal Energy Sensitivity Analysis Results

Scenario	**Energy Profile** [a]	**GW** [b]
1	NE (default)	184.6
	MA	-42.44
2a	NE (default)	206.7
	MA	-15.03
2b	NE (default)	209.6
	MA	-10.66
3	NE (default)	185.2
	MA	-16.25

[a] NE: Northeast U.S. energy profile; MA: mid-Atlantic U.S. energy profile
[b] GW: climate change impact in kg CO_2-eq.

4. Discussion: Using LCA to Inform Policy

4.1 Selecting the Best Food Waste Management Policy

Results indicate that diverting food waste from WTE to AD provides the greatest environmental benefits overall. Diversion to tunnel composting ranked slightly worse than WTE for overall environmental impact, and windrow composting performed worse than both of these scenarios. Because composting yields benefits that are difficult to quantify with LCA, such as weed suppression, increased soil productivity, and water conservation (Morris et al. 2014), the expected overall benefits of composting are believed to be underestimated. Additionally, because toxicity indicators were not included, the toxic effects of technologies were not accounted for. Generally, previous LCAs have determined that AD and composting have fewer potential impacts on human toxicity, human carcinogenicity, human respiratory effects, and ecotoxicity than WTE (Morris et al. 2013). Therefore, the benefits of AD and composting are also underestimated relative to WTE.

Diverting food waste to AD in Brookhaven provides the greatest potential for environmental benefit. Although tunnel composting ranked similarly to WTE when considering all indicators, the lack of ecotoxicity indicators and the inability to account for specific benefits of composting suggests that if these were included, diverting food waste to tunnel composting would result in greater environmental benefits than treating it with WTE. The business as usual scenario showed better environmental performance than the windrow composting scenario due to high energy requirements of this composting facility. It appears that diverting food waste to windrow composting would not provide environmental benefits overall based on the LCA results, although it is possible that if excluded impacts (ecotoxicty and composting benefits) were quantified and included, the net environmental benefit might favor windrow composting. So, composting may provide greater benefits than treating food waste with WTE, but these benefits are dependent on the specific composting technology used. However, all scenarios yielded greater environmental burdens than savings. This suggests that the best way to improve environmental performance is through food waste prevention. Waste prevention also eliminates upstream impacts of food production. The impacts of prevention are explored in chapter 6.

New England has included distance stipulations in food waste disposal bans so that food waste diversion is mandated only if a treatment facility is within a defined distance from the generator. These distance requirements may provide environmental benefit, but this is dependent on the tonnages of food waste diverted, and the specific impact categories of interest. The sensitivity analysis indicated that transport distance did not substantially impact global warming relative to overall system impacts. However, cumulative climate change impacts from transportation distance increases may be substantial. Other categories also may be affected more, particularly those where transport had a greater relative impact compared to other processes (depletion of fossil resources, marine and terrestrial eutrophication, and terrestrial acidification). Distance stipulations may reduce costs for longer distance transport of organics; they also represent a means of phasing in compliance as local transport and treatment capacity is developed.

The functional unit modeled here was 1,000 kg. of residential MSW collected curbside. In Brookhaven's residential waste districts, 154,555 tonnes of residual waste was disposed in 2014 and treated with WTE (1.34 tonnes/household/year); if about 14 percent of this waste was food waste, approximately 21,638 tonnes of food waste was incinerated (0.18

tonnes/household/year). This represents a substantial tonnage of waste that can potentially be diverted to alternative treatments, even if sorting efficiency rates are less than 100 percent. Therefore, environmental impact savings potentially achieved by diverting this food waste to AD or tunnel composting can be substantial. However, analyses indicated that the business as usual scenario performed as good as or better than alternative technologies in some impact categories. So, Brookhaven is currently performing reasonably well. In regions which rely on landfilling for residual waste treatment, or those with marginal energy profiles that have high environmental burdens, greater benefits may be gained from treating food with alternative technologies.

4.2 Baseline Technologies

The baseline scenario modeled here included all residual waste sent to WTE for energy recovery, which is Brookhaven's current practice. However, more than half of U.S. MSW is landfilled (USEPA 2014f). Although landfilling was not modeled here, prior LCAs indicate that it generally has higher environmental burdens than WTE or other alternative treatments (Guereca et al. 2006, Lee et al. 2007). Two reviews of food waste focused LCAs (Bernstad and Jansen 2012a, Morris et al. 2014), found that landfilling ranked as the technology with highest environmental burdens relative to other treatments (WTE, composting, AD) in multiple impact categories. Therefore, the relative benefits for food waste source separation with treatment through AD or composting will be greater when compared to landfilling.

4.3 Energy Portfolios

The marginal energy portfolio displaced from waste-derived electricity had a substantial impact on overall environmental impacts due to varying offset impacts. The potential effects of WTE or AD are partially dependent on the source of offset fuels so LCA findings will differ across regions with different marginal energy profiles. Here the substituted marginal profile was 81 percent natural gas, which has considerably fewer CO_2, NO_x, and SO_2 emissions than profiles in coal or oil dependent U.S. regions. Across the U.S., marginal CO_2 emissions vary from 486 kg/MWh (west) to 834 kg/MWh (midwest), SO_2 emissions vary from 0.2 kg/MWh (west) to 3.3 kg/MWh (mid-Atlantic), and NO_x emissions from 0.32 kg/MWh (west) to 1.07 kg/MWh (midwest) (Siler-Evans et al. 2012). Waste derived energy will show high impact savings when substituting for marginal energy in regions with high emissions; if it substitutes for renewable, non-polluting energy sources, perceived benefits are reduced. The benefits of waste derived energy substituting for fossil energy are likely to decrease in the future as more effective energy is captured from cleaner, non-fossil sources; the increase in fossil-derived plastic in waste streams, and organics reductions will also decrease the relative benefits of waste-derived fuels. As a result, the benefits from alternative food waste treatment will differ from one locale to another and can change over time. LCA can be important in identifying these disparities and driving effective policy design.

4.4 Other Important Factors for Decision Making in Waste Management

LCA is useful as a decision support tool for policy development because it indicates which conditions make a policy environmentally sound. However, there are unavoidable uncertainties and methodological limitations with LCA, particularly the subjective nature of outputs and the inability to account for social and economic factors. These limitations should be clearly identified to allow for policy makers to accurately assess the state of scientific knowledge. Thus, LCA can effectively drive policy without being an absolute tool for policy determination (Plevin et al. 2014). LCA results can be leveraged with other standard policy tools and analyses to provide a more complete understanding of options and impacts.

LCA has great value when selecting among waste management options. However, local circumstances, costs, and other factors not considered by LCA will impact the ultimate decision regarding waste treatment. Some of these additional quantitative and qualitative factors beyond those described by LCA include local environmental impacts (e.g., odor, noise), working environment factors (e.g., safety), investment costs, maintenance costs (Bernstad and Jansen 2012b), and stakeholder concerns (Garnett and Cooper 2014). Political goals (e.g. resource recovery, reduced emissions, energy recovery) will also affect which technological option appears to be the most beneficial. These factors were not assessed here. Separate management of food wastes is likely to cost substantially more than leaving food waste in the residual MSW stream due to separate collection and estimated tipping fees. However, if a municipality like Brookhaven implemented food waste diversion, it is likely food waste would be commingled with yard waste. This may reduce costs (there would be no additional collection routes, for instance) and would allow opportunities of scale. Chapter five presents a decision making framework to assist with analyzing the many factors that must be included when designing a waste management system.

4.5 Limitations and Directions for Future Research

LCA is increasingly used in waste management to assess the potential to reduce negative impacts to ecosystems, human health, or natural resources (Laurent et al. 2014b). Although it is a powerful and sophisticated analytical tool, it has methodological issues. It is difficult to make a direct comparison of waste treatment alternatives across studies (Lundie and Peters 2005, Bernstad and Jansen 2012a), and findings tend to be case specific (Vandermeersch et al. 2014). Functional units are often not equivalent, different modeling assumptions are made, output impact categories differ, different technologies are modeled, and modeling is conducted for diverse geographical areas. In a review of food waste LCAs, Bernstad and Jansen (2012a) found that LCAs had large differences in outcomes in absolute terms and in ranking of alternatives. They concluded that differences were due to variations in system boundary setting, methodological choices, and input parameters. Other reviews focused on organic and food waste LCAs also found substantial inconsistencies for waste treatment impacts from one study to another (Morris et al. 2013, Morris et al. 2014). Importantly, Villanueva and Wenzel (2007), in a review of paper waste management LCAs, determined that output differences were not due to differences in actual environmental impacts of the systems but to variations in methods and system boundary settings.

Many LCAs do not make the system boundaries and modeling assumptions clear (Cleary 2009), even though they have been found to drive LCA outputs (Winkler and Bilitewski 2007, Villaneuva and Wenzel 2007). Winkler and Bilitewski (2007) demonstrated the necessity of clearly indicating the scope and assumptions of an LCA in order to have confidence in LCA outputs, arguing for more transparency in LCA studies. Standards with specific instructions on LCA methods, including boundary setting and assumptions would increase comparability and transparency between LCAs. However, the complexity of waste LCAs and the site-dependent nature of some parameters seem to make strict standardization of LCA methods inappropriate. At this time, waste LCA findings are generally not comparable without substantial harmonization, and *post facto* harmonization may be unfeasible (Plevin et al. 2014). Still, it seems necessary to increase LCA transparency and overall quality so that a better understanding of the modeling approach can be achieved, facilitating interpretation of results, and allowing for potential cross-study comparisons.

Waste prevention is usually ranked as the most preferable waste management option. However, LCAs rarely assess waste prevention; it is difficult to quantify the benefits of waste that is not generated. Technically prevention alters the functional unit, thus making it challenging to compare results between scenarios (Ekvall et al. 2007). Waste prevention can liberate treatment capacity at disposal facilities. For WTE, this can result in higher energy value in residual waste due to lesser food waste. These effects are not typically accounted for in waste LCAs (Bernstad and Jansen 2012a), as most take a zero-burden approach. Upstream impacts, such as those from agricultural and industrial food production, may be substantial, and their inclusion is necessary for analysis of waste prevention effects (Oldfield and Holden 2014). The little work that has been done regarding waste prevention with LCA indicates that food waste prevention results in the highest environmental impact savings relative to other waste treatment approaches. Most benefits are gained from avoided food production (Gentil et al. 2011). Modeling food waste prevention for Brookhaven using LCA was beyond the scope of this study, but it is considered to be an important component of future work. The environmental impacts of upstream food production are explored in chapter six using the USEPA's simplified LCA software which includes approximations of prevention impacts.

A limitation of this study is that although it included a range of impact categories, there are still others that could have been examined, particularly with regards to human and eco-toxicity. These toxicity impacts are based on the relative risk and associated consequences of chemicals that are released into the environment. Characterization factors for toxic effects rely on models that account for a chemical's fate in the environment, human exposure, and toxicological responses (European Commission 2010). Another impact category that is absent from most LCAs is one that directly addresses soil quality and productivity, both of which provide critical ecosystem services (Morris et al. 2014). Here fertilizer replacement by compost was considered in the analysis, but other factors, such as water conservation and agricultural yield increase, were not. An analysis of soil benefits from food waste management options found that both aerobic composting and AD provided high soil-related benefits, including soil carbon sequestration, fertilizer replacement, water conservation, and yield increase (Morris et al. 2014). If these factors were all considered in the LCA, the performance of composting and AD relative to incineration would likely have been better. Capital goods associated with constructing waste facility buildings, reactors, and machinery were excluded, which is typical of most LCAs

(Brogaard et al. 2015). Extending LCA boundaries to include these data is an area for future research. Some of the inventory data were based on EASETECH defaults, and may not accurately represent exact conditions in Brookhaven and at treatment facilities on Long Island. Obtaining better, local inventory data which can be incorporated into models would be useful.

5. Conclusion

A life cycle assessment of the environmental impacts of four waste system scenarios was conducted for the Town of Brookhaven, NY, with a focus on food waste treatment. This approach allowed for the inclusion of local specificities to the model, such as waste composition and transport distances, and provided insight into potential approaches for improving the current waste management system with regards to food waste recovery. Analyses indicated that environmental burdens as a whole can be reduced by source separating food waste and treating it by alternative technologies (AD and tunnel composting). Results also indicated, however, that in some impact categories, the business as usual scenario performed better than the alternative technologies. This suggests that diverting food waste away from WTE will provide overall benefits (depending on the specific technology selected), but some environmental burdens will be increased. Therefore, other factors that influence decisions, such as cost, will be important factors for decision making. Sensitivity analyses indicated that the selection of marginal energy portfolios has considerable impacts on LCA results. This, combined with differing baseline technologies, demonstrates that food waste diversion may be considerably more beneficial in other regions, particularly those that landfill wastes and burn coal for electricity.

It is clear that selecting best waste management practices requires additional information and evaluation. The inability of LCAs to account for important parameters other than environmental impacts make them too one-dimensional to be used as a sole means to select waste treatments (Morris et al. 2014). Costs and stakeholder concerns must certainly be considered. A decision and evaluation framework for waste management systems was developed and described in chapter five to address these complexities. Emphasis was placed on using the framework to guide planning for systems targeting food waste management in chapter six.

Chapter 5. A Management Framework for Sustainable Solid Waste Systems

1. Introduction

In addition to having a good understanding of waste disposal drivers (chapter two), disposal quantities (chapter three), and the environmental impacts of technologies (chapter four), solid waste systems should be managed in a comprehensive, interdisciplinary manner which allows for incorporation of local concerns and evolution with changing situations and needs. This approach, which stems from Systems Theory, emphasizes that holistic approaches are important to implement effective changes within a system (Von Bertalanffy 1968). The prior dissertation chapters showed that food waste management is a complex, multi-faceted issue which crosses boundaries among environmental impacts, societal and ethical concerns, and economic factors. With this in mind, a framework for planning, implementing, and maintaining successful waste systems was developed which emphasizes sustainable decision making. This chapter describes the framework and how it can be used to improve waste management. Chapter six describes the application of the framework to food waste management, as well as its ability to address fundamental challenges with food waste prevention. The framework can assist with overcoming key obstacles encountered when establishing a food waste management program, and does so in a way which encourages success at various levels within a system. A theme of the framework is that effective waste management systems must successfully integrate knowledge from many disciplines, including engineering, science, policy, economics, sociology, and ethics, and be grounded in local conditions. This approach measures success across a range of indicators and enables these indicators to be tracked over time, which ultimately contributes to continual system improvements.

Because waste management has become increasingly complex in industrialized countries, there is an increased need for greater public engagement within the political and institutional decision making sectors (Garnett and Cooper 2014). The interdisciplinary framework developed here is important for solid waste management as existing waste management modeling and decision making approaches (e.g., life cycle assessment [LCA], cost benefit analysis, multi-criteria decision analysis) tend to focus solely on technological, financial, or environmental assessments and do not address the interdisciplinary nature of policy or the importance of social criteria, such as job creation or public perceptions (Skordilis 2004, Morrissey and Browne 2004). Methods for incorporating social aspects into waste system design are considerably less mature than methods for environmental and economic waste system assessments (Vinyes et al. 2013), and interdisciplinary approaches integrating all three aspects are even scarcer. Although the integration of public values with technical analysis are important for effective waste management (Garnett and Cooper 2014), existing waste models and decision making approaches tend to exclude the public from decision making processes and fail to consider all relevant stakeholders (Morrissey and Browne 2004). Governments generally oversee waste management, but their actions alone are far from sufficient to achieve sustainable waste management; rather, local level involvement is important, especially through engagement of the general public (Yau 2012). Also, existing approaches tend to lack clear definitions of system priorities (Joseph 2006).

These shortcomings of existing decision making approaches have led to calls for an improved methodology for sustainable waste management which integrates concerns of all stakeholders and includes detailed assessment at all stages of a system's progression (development, implementation, evaluation) (Morrissey and Browne 2004). There currently is no consensus on the best way to integrate methods for sustainability assessment, since such an assessment is dependent on the purpose of analysis and specific local factors (Jeswani et al. 2010). The objective of this work was to develop a framework which fills these gaps and facilitates waste system management. This framework is proposed as a means to address the need to integrate existing waste models (e.g., LCA, cost-benefit analysis) in a way which accounts for overall sustainability concerns and emphasizes social priorities to enable better decision making for sustainability. This framework was designed to be broad enough to allow for easy integration of local knowledge and approaches, as well as project specific concerns, thus facilitating its incorporation into extant waste management structures.

2. An Improved Framework for Waste Management

Sustainable waste management systems are environmentally effective, economically affordable, and socially acceptable (Nilsson-Djerf and McDougall 2000). So, systems must relate to local environmental, economic, and social priorities and encourage stakeholder and public engagement in decision making (Garnett and Cooper 2014, Joseph 2006, Petts 2000).

Such complex management situations are better handled if they are supported by tools for evaluating overall system performance which integrate these concerns (Coelho and Moy 2003). However, traditional waste planning models focus solely on technological, financial, or environmental systems; few refer to the interdisciplinary nature of policy, and most do not analyze social criteria (e.g., employment, social acceptance) (Skordilis 2004).

An interdisciplinary systems management tool (referred to as the framework) was developed to facilitate the planning, implementation, and maintenance of sustainable waste systems. It aims not to replace existing decision making approaches, but rather to enable their integration to allow for inclusion of overall sustainability concerns. It defines key considerations for designing waste systems and describes how to monitor system performance over time. The framework was developed based on knowledge of waste systems and assessments, current data needs, an examination of challenges impacting waste systems, and areas where success has been observed with regards to waste policy in the past. Successful implementation of this framework is based not only on waste diversion rates or economic criteria, but also on stakeholder engagement, fulfilment of social priorities, and other concerns. This framework was designed for municipal solid waste management, although it may be extended to other waste systems, such as industrial wastes.

2.1 Comparison to ISO 14001

The framework has some similarity with the ISO 14001 standard for the development, implementation, and maintenance of environmental management systems (EMS) developed by the International Organization for Standardization (ISO). The ISO 14001 standard has been adopted by a range of organizations since its creation in 1996, primarily in Europe and Asia (To and Lee 2014); almost 286,000 organizations worldwide have an ISO 14001 certified EMS (ISO 2013). The standard provides a procedure for any type of organization to develop and maintain an EMS. It includes 18 elements: establishment of EMS scope; environmental policy; environmental aspects and impacts; legal and other requirements; environmental objectives, targets and programs; resources and responsibility; competence and training; communication; documentation; control of documents; operational control; emergency preparedness/response; monitoring and measurement; evaluation of compliance; corrective and preventative actions; control of records; internal auditing; and management review (ISO 2004). The effectiveness of the standard at reducing environmental impacts is unclear; some studies have found it to be beneficial, while others indicated that ISO 14001 certified organizations showed no environmental improvement (Comoglio and Botta 2012, Zobel 2015).

The major similarities between ISO 14001 and the waste framework are that they define an approach to manage systems, including defining objectives, setting targets, and monitoring improvements. A desired outcome of the framework is to allow for continual improvement in system performance over time, shared with ISO 14001. Another similarity is that both approaches are general; they are meant to be adapted by a variety of organizations, ranging in size, function, and purpose.

Besides similarities in the overall structure of the framework and ISO 14001, specific details differ considerably. The framework is designed specifically for waste management systems, through integration of waste specific concerns. Also, instead of focusing solely on environmental performance, the framework encourages an interdisciplinary sustainability focused approach which integrates environmental, economic, and social factors. The framework is less rigid than ISO 14001 to allow for easy integration of local knowledge and approaches, thus facilitating its incorporation into pre-existing waste management systems. Unlike ISO 14001, the waste framework does not involve tedious and time consuming practices (Westly and Rogoff 2008). Instead, it aims to be relatively easy and quick to implement, making it time and cost effective.

An issue with waste management systems is that managers often lack the resources and time to implement complex management approaches. A summary of the experiences of four waste organizations that implemented an ISO 14001 EMS found that large expenditures of time and money were required to implement and maintain the systems (Westly and Rogoff 2008). Kent County, MI, reported that their expenses were approximately $25,000 to implement an ISO-14001 EMS for their landfill, and significant time from county employees was required, including at least 10 to 20 hours from a dedicated environmental compliance manager. In King County, WA, the implementation costs for a waste facility EMS were between $44,000 and $72,000 annually for three years. It required about 1,000 to 1,200 personnel hours per year for three years, and maintenance required 200 to 250 personnel hours per year (Westly and Rogoff 2008). A resource intensive EMS is not ideal for many waste systems, particularly due to limited resources. Furthermore, a generic EMS does not account for concerns specific to waste management and it does not incorporate economic or social factors.

2.2 Framework Objectives

The framework provides guidance on the implementation of sound waste management practices which helps ensure that important system aspects are not overlooked when planning or maintaining a waste management system. In fact, some framework aspects are already conducted to some degree in waste planning, but the framework helps ensure that all key aspects are acknowledged, and that the system is continually monitored over time. The overall goal of the framework is for it to serve as a practical tool for the application of sustainable programs for waste management. Programs can include specific technology or policy options. Specific objectives of the framework are:

1. Allow for system components to be well-defined
2. Maintain compliance with applicable regulations
3. Integrate environmental, social, and economic concerns into waste systems
4. Enable data collection and performance assessment
5. Allow adjustments to be made over time for improvement

2.3 Framework Components

There are four overarching components of the framework: Plan, Implement, Evaluate, and Improve (Figure 22, Table 32). The purpose of the first overarching principle, Plan, is to encourage municipalities to clearly define overall system objectives and to identify what programs are necessary to achieve them. A key aspect of the Plan component is that the regulatory and financing structures of the system must be clearly defined, as well as the population targeted by waste system policies. The Plan component also emphasizes stakeholder outreach, which aims to improve stakeholder relations and to leverage their expertise. The Plan stage is the most important part of the framework as it encourages planners to think through many of the key components of the waste system in light of overall objectives. By starting the framework by defining overall objectives, managers can integrate these objectives throughout the whole system as they work through the steps of the framework.

The next overarching principle, Implement, refers to the daily operations of the system. A key aspect of this component is defining targets and performing regular data collection to assess progress towards objectives and targets. An issue with many solid waste systems is the lack of accurate and complete data. The framework aims to address this issue by encouraging regular, comprehensive data collection. The purpose of the Evaluate principle is to evaluate system performance and to critically analyze challenges that have been experienced. This is an important step, especially after the framework has been implemented for some time to determine if system objectives are being achieved. The final overarching principle is Improve. This principle encourages frequent review of the system and its performance, and modification if necessary. Modification of aspects of the Plan stage should be done as necessary (e.g., update legal requirements if new laws are passed). The framework is intended to be used continuously so that systems are repeatedly evaluated and improved.

Figure 22. The Four Principles of the Waste Management Framework

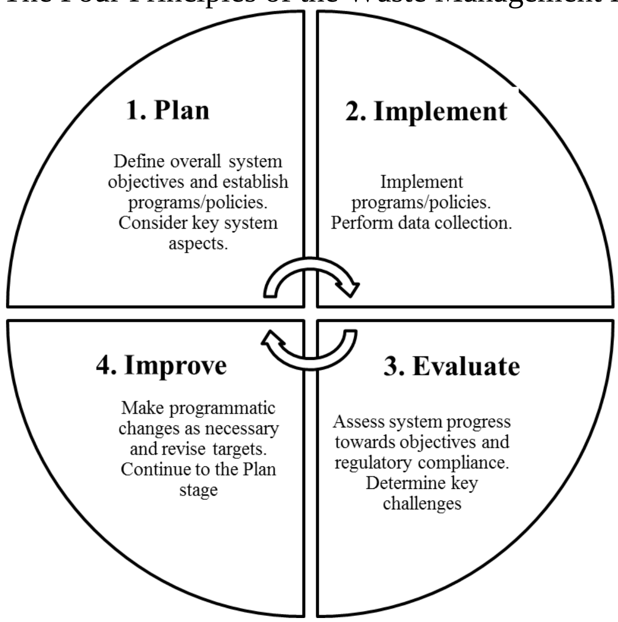


Table 32. Management Framework for Waste Systems

Overarching Component	Step	Description
Plan	1. Define system	a. Define scope of system (e.g., scale, time-frame) b. Define system boundaries c. Define overall system objectives, including environmental, social, and economic objectives. Integrate local concerns d. Clearly state definitions for key terms
	2. Programs and Policies	a. Determine programmatic (including technological and policy) options for achieving overall objectives b. Evaluate the program with regards to Steps 3-8 of the framework
	3. Requirements	a. Identify and/or define applicable legal requirements b. Identify and/or define other applicable requirements (e.g., institutional)
	4. Resources	a. Define required economic resources; consider long term funding b. Define other required resources (e.g., human resources, specialized skills) c. Ensure required resources are available. Perform detailed

		cost assessments, such as cost benefit analysis, if necessary
	5. Responsibilities	Define roles and responsibilities for system managers, other personnel, and stakeholders
	6. Environmental Impacts	Evaluate environmental impacts of program. Use LCA or another comprehensive approach if possible
	7. Stakeholders & Social Impact	a. Identify stakeholders and their concerns regarding the system b. Define methods for stakeholder communication, including regular outreach and education; include approaches for integrating their knowledge and concerns early in the planning process c. Identify impacts of program on society (e.g., job creation)
	8. Measure	a. Identify and define performance indicators which are measureable and consistent with the overall system scope and objectives; include environmental, financial, regulatory, social, and stakeholder concerns (as identified during previous planning steps) b. Define methods for ensuring sufficient and regular data collection
	9. Select Program/Policy	Select the best program option based on findings from Plan steps
Implement	10. Targets	a. Identify specific targets based on the indicators selected in Step 8 b. Define the means in which target will be achieved and set time-frame for achievement
	11. Implement program	Implement the program
	12. Collect data	Collect data according to plans outlined in Step 8
Evaluate	13. Evaluate progress	a. Determine if overall system objectives and specific targets are being achieved (as identified in Steps 1 and 10) b. If achievement is not reached, identify reasons why
	14. Evaluate compliance with requirements	Determine if compliance with requirements (as identified in Step 3) is achieved
	15. Challenges	Identify challenges observed within the system
Improve	16. Programmatic changes	a. Identify ways to improve existing programs, especially if targets are not achieved b. Plan and implement new programs if necessary (use Steps 1-9)
	17. Revise Targets & Continue	a. Revise targets based on current performance b. Modify other components of the Plan stage as necessary c. Continue following the framework to repeatedly evaluate the system allowing for continual improvements

The framework may be applied to a whole system (e.g., whole waste management system) or to a subsystem (e.g., food waste management system). It may be used at various system stages; it can be first used as a planning tool to design or decide on a new program, policy, or technology and then to evaluate outcomes, or it can be used to evaluate an existing program, and then make improvements. It can be used to prevent waste or to manage waste effectively after it has been generated. The framework should be followed in order and documentation for all steps should be performed. However, unlike the strict, time consuming, and costly documentation requirements of ISO14001, the framework allows managers to perform documentation and document control in any manner they choose, thus facilitating its integration with current practices. It is possible to perform the Plan steps multiple times if there are various programmatic options being considered. This facilitates the comparison between options and the eventual selection of the best option. If this approach is utilized, once the best option is selected, the rest of the framework (Implement, Evaluate, Improve) should be followed. The framework is purposely general so that it may be utilized by a range of waste management systems and local considerations and situations can be incorporated.

2.4 Key Framework Aspects

A major aspect of the framework is the integration of diverse stakeholders into waste planning, implementation, maintenance, and evaluation. This involvement not only allows for reflection of concerns and interests of stakeholders, but also extends the knowledge base for decision making (Garnett and Cooper 2014). National, state, and local governments, technical experts (e.g., academics, consultants), legal representatives, funding agencies, community groups, media, industry, and the general public play major roles in supporting waste policy actions. Their inclusion facilitates effective planning, particularly concerning waste facility siting, which often if one of the most difficult parts of waste system design. Identifying stakeholders and their interests is necessary to ensure their participation and involvement in waste management (Joseph 2006). It also is important that regular stakeholder communication be conducted over time. Table 33 outlines key stakeholders and examples of their roles in waste management.

Another key aspect of the framework is the collection and monitoring of data to assess system performance. There often are insufficient data and metrics in waste management which restrict complete policy evaluation. This dissertation aimed to close some of the data gaps with regards to food waste, although there still needs to be considerable improvement in waste data as movement is made towards more sustainable waste management. A successful management system involves stipulations for comprehensive data collection which enables waste managers to assess system performance. Complete, accurate data enable quantitative-based policy making and target setting. Furthermore, increases in the number of well managed waste systems with complete data collection will improve the overall data situation for U.S. waste systems as a whole. This enables managers to compare performance from one system to another, and to learn from successes and failures of others.

Table 33. The Roles of Stakeholders in Sustainable Waste Management

Stakeholder	Examples of Roles
National & state government	Set environmental regulations; support local municipalities; provide education to local government; assess innovation
Local government	Monitor system performance; drive public education; set targets/objectives; ensure availability of adequate human & financial resources; provide infrastructural inputs & services; enforce and comply with regulations
Technical experts	Determine which technologies & policies are most effective; conduct assessments, such as LCA or cost-benefit analyses; drive innovation
Policy makers	Develop polices
Social services	Address social concerns, including job creation & environmental justice
Funding agencies	Support/fund projects
Legal representatives	Develop legal regulations; ensure adherence to legal requirements; prepare contracts
Community groups	Promote local concerns; participate in advisory groups
Media	Contribute to environmental awareness; inform the public and educate about major issues
Waste and transportation industries	Mange wastes as dictated by policy and regulation; drive innovation
General public	Participate in decision making regarding effective programs; adhere to waste policies; pay for waste services; assist in identifying sites for waste facilities; work in waste management facilities

Part of the data collection process involves collecting sufficient data to examine performance indicators. Performance indicators should be selected during the Plan stage based on relevant environmental, social, and economic concerns. So, in addition to including indicators focused directly on managed wastes, it also is important to incorporate indicators that address other sustainability issues such as waste prevention, public education programs, affordability of programs, and extent of stakeholder engagement. Wilson et al. (2015) support the use of performance indicators for waste systems that extend from physical and technological system components to sustainability aspects (social, institutional, political, financial, economic, environmental, technical), and stakeholder concerns. Greene and Tonjes (2014) defined evaluation criteria for waste management system performance indicators (Table 34). These criteria should be included when deciding on indicators using the waste management framework.

Table 34. Waste System Performance Indicator Evaluation Criteria

Criteria	Definition
Direct	Indicator measures closely to the possible result it is intended to measure
Objective and Specific	No ambiguity in measurements; indicator is clearly defined and uses common definitions
Clear	Indicator should be simple and easy to interpret
Practical	Data can be obtained timely at reasonable costs
Reliable	Data for indicator is of sufficient, dependable and consistent quality for decision making
Useful for Waste Managers	Indicator provides meaningful measurement of system change; indicator is useful for daily decision making regarding system; indicator indicates progress towards improved system design
Relevant	Indicator provides information that is of priority interest; indicator is important for communicating information about systems

Adapted from Greene and Tonjes 2014

2.5 Guidance on Using Framework for Planning

Table 35 provides guidance on key considerations for decision makers when using the framework for waste system planning. It demonstrates the questions that should be addressed when selecting among policy or technology options. It is possible to set up a decision matrix using the framework as a guide to systematically and quantitatively compare one option with another. The table is not meant to be an exhaustive list of considerations, but is provided to demonstrate how the framework can guide decision making.

Table 35. Key Considerations

Step	Questions to Consider
1. Define system	• What is the scope of the project? • What system objectives do you want to achieve? • What is the overall timeline for the project?
2. Programs and Policies	• What policy options are under consideration? • Which policy option aligns best with system scope and objectives? • Does the policy allow for changes to be made to it over time?
3. Requirements	• Does the policy align with existing legal and other requirements? • If new regulation is required, is it feasible to implement within the existing regulatory environment? • Is there a way to ensure compliance with the policy? If so, how?
4. Resources	• What is the financial cost to implement and maintain the policy? • What human resources (e.g., staff time) are required

	to implement and maintain the policy? • Are there any other specialized resources that are required? • Are the required resources available? If not, how will you ensure that they are available? • Is the infrastructure required for the policy in place? • Are there means to facilitate public compliance with the policy (e.g., economic incentives, technical assistance)?
5. Responsibilities	• What roles and responsibilities are required for managers? Is this feasible? • What roles and responsibilities are required for other personnel? Is this feasible?
6. Environmental Impacts	• What are the environmental impacts of the policy? • What are the main factors that affect the environmental impact of the policy (e.g., travel distance of waste to processing facility)? • What local environmental issues are of concern?
7. Stakeholders & Social Impact	• Who are the main stakeholders? • How will you reach out to stakeholders and incorporate their concerns into the policy? • What are the concerns of the main stakeholders? • Who will be in favor of the policy? Why? • Who will be in opposition of the policy? Why? • Will the policy provide social benefit (e.g., job creation)?
8. Measure	• Do performance indicators reflect system objectives? • Do performance indicators address issues from various sustainability issues (economic, social, and environmental)? • Are the performance indicators clear, specific, practical, reliable, useful, and relevant? • Are means for data collection feasible? • Is there room to improve the system over time? • What are the expected obstacles to implementing and maintaining this policy in the short-term and long-term?
9. Select Program/Policy	• Which program/policy: - aligns best with your system? - enables system objectives to be achieved? - is acceptable to stakeholders? - is feasible to implement within timeframe?

3. Conclusion

Effective solid waste policies and programs need to be planned carefully, with consideration for diverse factors, including regulatory requirements, financial needs, environmental impacts, and social implications. The waste management framework helps ensure that key factors are considered when developing solid waste policies and programs. Furthermore, stakeholder engagement, communication, and a fostering of their understanding of policies are important for success. When the public and stakeholders are well-informed about policy options, the importance of initiatives, and pathways for participation, better decisions and outcomes will result. Waste systems also need sound data collection and performance evaluation processes to allow for improvements over time. Comprehensive data collection combined with well-defined indicators is necessary. Indicators should be selected based on local concerns and initiatives.

The waste management framework fosters waste system planning, implementation, and maintenance across separate disciplines to facilitate the management of complex systems. The framework has numerous benefits. It is easy to implement and allows for the integration of existing waste assessment approaches, as well as local conditions. Initial implementation of the framework should be carefully analyzed to determine exactly how it helps system performance, and to identify areas where the framework may be improved. The framework is applied to food waste management next in chapter six as an example. This emphasizes the benefits that can be achieved by using the framework, particularly how it helps minimize challenges typically faced in waste management.

Chapter 6. Motivators and Challenges to Food Waste Prevention

1. Introduction

Analyzing disposed waste provides insight into the steps that can be taken to improve waste management beyond current programs. This information serves as a starting point to develop new or improved waste policies and system structures aimed at diverting waste away from disposal. The prior dissertation chapters provided information which supports an informed discussion on ideal food waste management. The importance of studying food waste and generation drivers were first reviewed to provide an understanding of why societies dispose so much food waste, and the associated environmental, economical, and social impacts. This provides rationales for reducing waste generation. Next, the amount of food waste disposed as municipal solid waste (MSW) was quantified, showing that food waste makes up a considerable portion of the waste stream. This justifies targeting food waste for prevention and recovery. Chapter four demonstrated the importance of selecting waste treatment technologies that minimize environmental impacts. All modeled food waste treatment approaches were found to have greater environmental burdens than savings. It underscored that careful consideration is required in decision making, as environmental impacts vary with selected technologies and local circumstances. Chapter five highlighted that waste management planning covers a wide set of interdisciplinary issues. A framework for developing, implementing, and maintaining sustainable waste programs that addressed these concerns was proposed.

The lack of mainstream attention and effort to address food waste suggests major changes are needed to drive food waste reduction (Finn 2014). Because food waste is such a complex issue, crossing environmental, social, and economic bounds, it is necessary to adopt a multi-pronged approach to address it that integrates various sustainability initiatives. This chapter discusses ideal food waste management approaches based on the findings from previous chapters, beginning with the potential environmental, social, and economic benefits of food waste prevention, followed by recommendations for planning and implementing a food waste prevention program. The waste management framework (described in chapter five) can be an effective tool for designing sustainable food waste management policies since it encourages multi-faceted approaches to waste planning. Here key challenges to food waste prevention and means to address them were assessed.

2. Motivators for Food Waste Prevention

Efforts are needed to actively prevent food waste disposal (waste prevention); waste prevention is the best way to achieve environmental, economical, and social benefits (UNEP 2014). After prevention efforts are exhausted, food waste remaining in the disposal stream may be diverted to alternative treatments (waste diversion) to capture nutrients and energy. Definitions of key terms as used in this chapter are given in Table 36.

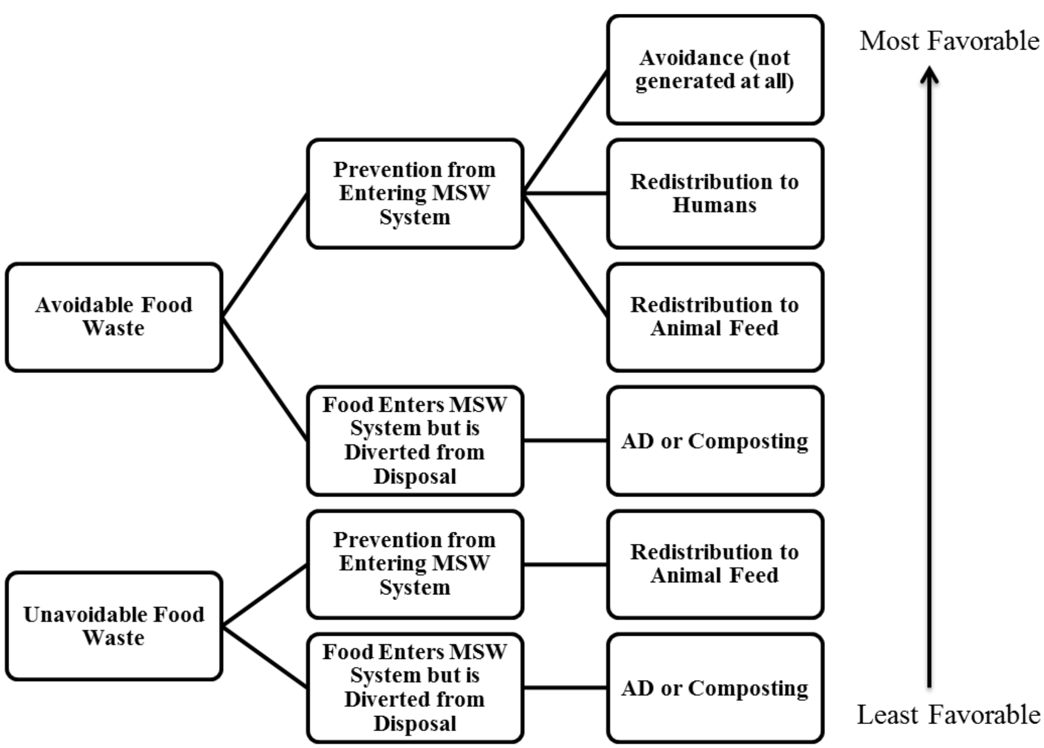
Table 36. Definition of Key Terms

Term	Definition
Waste avoidance	Wastes which are not created at all (e.g., consuming food before it is allowed to spoil, thus avoiding wastage)
Waste prevention	Wastes which are avoided and wastes which are prevented from entering MSW systems, such as through redistribution to feed humans or animals
Waste diversion	Wastes which enter MSW systems, but are treated by alternatives to disposal means, such as anaerobic digestion (AD) or composting.
Waste disposal	Wastes which are treated with waste to energy incineration (WTE) or landfill
Avoidable food waste	Food that is discarded because it is no longer wanted or has been allowed to age past its best (WRAP 2010)
Unavoidable food waste	Discarded food that generally is not considered edible in normal situations, such as apple cores and meat bones (WRAP 2010)

Generally, waste management policies separate waste prevention from diversion from disposal. It is common for waste managers to focus little on waste prevention, particularly because once waste reaches treatment facilities, it can no longer be prevented (Wilts 2012). The substantial global impact of food waste suggests rethinking this general approach, as it blocks comprehensive systems-approaches to sustainability in waste management. Close linkages between waste prevention and diversion require that they be considered together systematically to evaluate how one affects the other. Waste managers should work to actively address prevention as a component of their waste programs, and leverage these initiatives to complement existing and future diversion programs.

Figure 23 illustrates preferred food waste management options; avoidance of food waste is the most favorable option. If food waste cannot be avoided, effort should be made to prevent it from entering the MSW system by redistributing it to feed people or animals. Chapter four showed that all modeled food waste treatments had environmental burdens; waste prevention reduces these burdens by lessening resource and energy consumption to process the waste, and minimizes waste management costs (Bartl 2014). Donating still-edible food for human consumption helps solve societal problems, as it assists the needy and reduces the environmental impacts of food that is produced but not consumed. Food banks and rescue programs which distribute food through charitable donation and serve the hungry were established throughout the U.S. in the 1960s. An obstacle to food donation is that some food waste may be unfit for human consumption, or may be deemed to be unfit under safety and hygiene regulations, and logistical barriers to donation may be substantial (Schneider 2013b); logistical barriers may be overcome to some degree with strong coordination efforts. Food that cannot be donated may be managed by other means, such as by feeding it to animals. If that is unfeasible, it can be treated through preferred MSW technologies. Chapter four indicated that anaerobic digestion (AD) has fewer environmental burdens than composting. These options are both favored over sending food waste to disposal; if disposal is necessary, treatment with WTE is preferred over landfill (as described in chapter four).

Figure 23. Preferred Food Waste Management Hierarchy



Home composting was excluded from Figure 23 primarily due to lack of reliable data and assessments, making it difficult to accurately determine its impacts. Some work suggests its benefits may be location specific, so its use cannot be generalized as universally beneficial. Andersen et al. (2012) found that home composting performed better than or as good as WTE and landfilling, but the analysis did not include centralized composting or AD so it is unclear whether home composting provided benefits over these approaches. Martinez-Blanco et al. (2010) found that home composting can accompany centralized composting in areas with low population densities where home composting is feasible.

2.1 Social and Economic Benefits of Food Waste Avoidance

Food waste avoidance can reduce social and economic harms of food waste. A British Government study found that reducing food waste across the entire food chain will be a key part of any strategy to sustainably and equitably feed the world's growing population (Foresight 2011). The United Nations Environment Programme (2014) stated that minimization of food waste and loss are essential for achieving global food security because it will improve the overall availability and supply of safe and nutritious food for human consumption. Essentially, food waste avoidance in one region could lead to a higher availability of food elsewhere (Gentil et al. 2011). Stuart (2009) argues that food waste reduction in the developed world will liberate agricultural land and resources for other uses, such as growing food for those that do not have enough. The economic impacts of food wastage are substantial and reducing waste can lessen these impacts. A global estimate by the FAO (2013) estimated that the direct economic cost of food wastage of agricultural products (excluding seafood) (based on producer prices only) was approximately $750 billion (U.S.).

2.2 Environmental Benefits of Food Waste Avoidance

Avoidance of food waste has the highest environmental benefit relative to other waste management approaches due to the reduced environmental impacts gained from avoided food production (Gentil et al. 2011, Schott and Andersson 2015). At a global level, Kummu et al. (2012) estimated that one quarter of the freshwater, and one fifth of the cropland and fertilizers are used to produce food that is lost or wasted. In the U.S., the production of wasted food requires the expenditure of over 25 percent of the total freshwater used in the U.S., about 300 million barrels of oil (Hall et al. 2009), and represents two percent of annual energy consumption (Cuellar and Webber 2010). Schott and Andersson (2015) found that preventing avoidable food waste resulted in greater potential reductions of greenhouse gas emissions than if the waste was treated with WTE or AD. They concluded that greatest climate change impact savings are achieved when preventing food waste first, then treating the remaining food waste through WTE or AD. Here, the United States Environmental Protection Agency's (USEPA) WARM model (Microsoft Excel Version 13) was used to demonstrate potential environmental benefits from food waste avoidance and illustrate how avoidance achieves greater benefits than food waste diversion or disposal.

2.2.1 Background on WARM

WARM is a simplified, stream-lined life cycle assessment (LCA) developed specifically for waste managers that has fewer outputs than other waste LCAs, such as EASETECH. WARM provides approximations of greenhouse gas (GHG) emissions and energy savings for baseline and alternative waste management practices, including source reduction, composting, incineration, landfilling, and recycling (it does not include AD). It recognizes 50 material types, including several groups of mixed materials, and has been used in previous waste-focused studies (Chester and Martin 2009, Vergara et al. 2011, Greene and Tonjes 2014). USEPA (USEPA 2014b) has described WARM, including underlying assumptions and emission factor calculations.

The 13th version of the model (released June 2014) is improved from earlier versions, particularly with regards to food waste. New material categories were added to the model, and the food waste material was modified to include three categories: grains, fruits and vegetables, and dairy. Each category has its own level of detail and associated emissions, including those associated with upstream effects (the energy required to grow, process, and transport the food). These impacts are counted if the food is wasted. The new version includes improved emission factors for composting and landfilling, including revised methods for estimating landfill gas emissions and the inclusion of fugitive gas emissions from composting. An important change to the model is that landfilled food waste is considered more likely to generate methane than other organics, such as office paper. The model also now accounts for the faster decay rates of food waste compared to other materials, which indicates that methane from food waste is more likely to escape from landfills before gas collection systems are installed (USEPA 2014b).

2.2.2 Modeling approach and assumptions

Three baseline and alternative scenarios were modeled to assess the impacts of food waste avoidance compared to waste treatment by composting, landfilling, and WTE. The functional unit was 100 tons of food waste. The food waste category is a mixed material, containing 16 percent grains, 61 percent fruits and vegetables, and 22 percent dairy. Three sets of baseline and alternative scenarios were modeled, one for each waste technology (landfilling, incineration, composting). It was assumed that in the baseline scenario, all food waste was treated with a waste technology, and in the alternative scenario, 60 percent (60 tons) of the food waste was avoided (based on the assumption that 60 percent of food waste is avoidable as determined in a study by WRAP [2013]), and the remaining 40 percent (40 tons) was treated with the baseline waste treatment technology. The scenarios were:

1. Baseline: landfilling (100 tons); Alternative: landfilling (40 tons) and avoidance (60 tons)
2. Baseline: incineration (100 tons); Alternative: incineration (40 tons) and avoidance (60 tons)
3. Baseline: composting (100 tons); Alternative: composting (40 tons) and avoidance (60 tons)

WARM incorporates regional marginal electricity grid factors to calculate avoided electricity-related emissions; WARM's national average category was selected. WARM varies outputs based on landfill gas recovery; national average, which is based on the estimated proportions of landfills in landfill gas control in 2012, was used. Default distances (20 miles) were selected as the transport distances for wastes from collection point to treatment facility.

2.2.3 WARM results

In all three scenarios, the alternative scenario performed considerably better environmentally than the baseline (Tables 37 and 38). This indicates that food waste avoidance is the best way to prevent GHG emissions and save energy. The greater reductions observed for avoidance relative to composting substantiate that priority should be first given to preventing food waste, and then, once prevention initiatives have been exhausted, treating waste with composting. Greater GHG reductions and energy savings were also achieved by treating food waste with incineration rather than landfilling, even with landfill gas recovery.

Table 37. GHG Emissions from Baseline and Alternative Scenarios
(Functional Unit: 100 tons food waste)

	Baseline ($MTCO_2E$)	Alternative ($MTCO_2E$)	Total Change (Alternative-Baseline) ($MTCO_2E$) [a]
Landfilling	75.5	-15.3	-90.7
Incineration	-12.1	-50.3	-38.2
Composting	-15.3	-51.6	-36.3

[a] A negative value represents an emission savings

Table 38. Energy Savings from Baseline and Alternative Scenarios
(Functional Unit: 100 tons food waste)

	Baseline (million BTU)	Alternative (million BTU)	Total Change (Alternative-Baseline) [a] (million BTU)
Landfilling	33.6	-418.4	-452.0
Incineration	-207.5	-514.9	-307.3
Composting	58.4	-408.5	-466.9

[a] A negative value represents an energy savings

3. Planning and Implementing a Food Waste Prevention Policy

Waste prevention has been studied much less frequently than waste diversion and treatment. Due to the inherent difficultly in studying and implementing waste prevention programs, there has been little quantitative work assessing its environmental impacts (Gentil et al. 2011). Little research has directly addressed factors that motivate, enable or inhibit food waste prevention behaviors (Graham-Rowe et al. 2014). Nevertheless, the potential benefits are large because prevention can solve multiple issues across a range of disciplines. Currently in the U.S. there is no widespread or visible political or social momentum to prevent food waste and prevention initiatives are rare (Buzby et al. 2011, Finn 2014).

In the developed world the greatest potential for food waste prevention is with retailers, food services and consumers (Parfitt et al. 2010, Papargyropoulou et al. 2014). Consumer and foodservice food waste is the largest source of food loss and waste in the U.S. and much of this food waste is avoidable. Understanding the degree to which food prevention is feasible is important for designing strategies to minimize food waste (Papargyropoulou et al. 2014). However, estimations of the proportion of food waste that is avoidable differ considerably across studies; estimates for the proportion of avoidable food waste range from 34 percent in Sweden (Schott et al. 2013), to 47 percent avoidable and 18 percent partially avoidable in Germany

(Kranert et al. 2012), to 60 percent avoidable in the U.K. (WRAP 2013). More studies documenting the proportion of disposed food waste that is avoidable would be beneficial, especially in the U.S. where data are lacking.

3.1 Policies to Prevent Food Waste

Waste prevention requires changes in people's behavior, both collectively (e.g., companies) and individually (Wilson 1996). The European Union has identified behavior changes by public authorities, citizens, and businesses as important for food waste prevention (BioIntelligence Service 2011). Analyzing behavioral causes for food waste generation is the first step in planning an approach to change people's actions to reduce food waste. There is an array of consumer attitudes, values and behaviors towards food which contribute to the propensity to waste food (Parfitt et al. 2010). Buzby et al. (2011) emphasize that consumers have different preferences and attitudes, indicating the drivers for food waste prevention may differ from person to person. National circumstances and cultural norms have also been linked to food wastage (BioIntelligence Service 2011). Evans (2012) found that wastage was a consequence of social and material conditions, rather than specific individual thoughtlessness.

Policies for food waste prevention should target the circumstances and actions that lead to food wastage, and should be informed by motivations for waste production. Chapter two discussed the variety of causes for retail and consumer food waste, which include: overstocking; over or inadequate food preparation (Quested and Johnson 2009); lack of food preparation skill; defects in food or food packaging; spoilage/food not being used in time; routine kitchen preparation wastes; confusion over food labels (Kosa et al. 2007); and misconceptions regarding food safety (Pearson et al. 2013). There also generally appears to be a lack of concern and awareness regarding food waste (Buzby et al. 2011) and a perception that food waste prevention is not a priority (Graham-Rowe et al. 2014). Table 39 lists prevention initiatives based on behavioral factors (Table 39).

There are a range of policy options to support food waste prevention (UNEP 2014); specific policy options to prevent food wastage may be applied to residential, commercial, and institutional sectors (Table 40). Ideal food waste prevention policies should address multiple prevention mechanisms simultaneously because prevention is not created by one, but by many behaviors (Cox et al. 2010). Policy packages should also address concerns people have about food wastage. These include: (1) food waste is a waste of resources (money and edible food); (2)

wasting food is wrong and yields feelings of guilt (Graham-Rowe et al. 2014); and (3) food waste negatively impacts the environment (Doron 2013).

Table 39. Initiatives to Prevent Food Waste Based on Behavioral Factors

Behavioral Factor	Description	Initiatives to Prevent Waste
Over Preparation/ Large Portion Sizes/Undesired Food	Excess food that is prepared but that is not consumed (includes plate waste)	1. Public/employee education regarding proper food preparation, portion sizes, and on importance of ordering flexibility to ensure people like the food they are served 2. Food redistribution policies for edible retail and commercial food (e.g., to a food bank)
Inadequate Food Preparation / Lack of Food Preparation Skill	Food that is prepared incorrectly (such as by burning) or poorly (such as food that does not taste good) which results in wasting; food that is wasted due to an inability to reuse excess food or incorporate left-overs into a new meal	Public/employee education regarding proper food preparation and reuse
Defects in Food or Food Packaging	Food that is disposed due to imperfect qualities of the food (such as bruising) or damaged food packaging (includes out-grading)	1. Logistic improvements (e.g., improved transportation that reduces food damage) 2. Food redistribution/donation policies for edible retail and commercial food (e.g., to a food bank)
Over Stocking	Excess food that is purchased but not consumed /sold (either at consumer or retail levels)	1. Public/employee education regarding food purchasing and planning 2. Logistic improvements (e.g., stock management improvement for retailers, reduction in bulk discounts)
Spoilage/Food Not Used in Time/Confusion Over Date Labels	Food that is allowed to spoil before it can be consumed/sold or food that is believed to be inadequate for consumption based on personal preferences, date labels, or conceptions about food safety	1. Public/employee education regarding food storage, food safety, and food planning 2. Improved, easily understandable food labeling systems 3. Logistic improvements (e.g., stock management improvement for retailers, improved product packaging)
Routine Kitchen Preparation Wastes	Non-edible food components that are disposed of as part of routine kitchen preparation (e.g. apple cores)	These wastes are difficult to reduce; therefore, they are best targeted with policy options for MSW systems, such as food waste diversion policies to encourage AD or composting
Lack of Awareness or Concern About Food Waste	Lack of awareness or concerns about wasting food	Education regarding food waste and why it is an environmental, economic, and social concern

The package of prevention policies should encompass three key aspects: Values, Skills, and Logistics. The first aspect, Values, involves addressing values and perceptions which drive behavior. An example of a Values policy is one which teaches people about the importance of environmental and social altruism, and how preventing food waste can provide benefits (Wilson 1996). The next policy component, Skills, enables people to change their behaviors, such as by providing training on how to prevent food waste. The final aspect of a policy package is

Logistics. These policies facilitate the prevention of food waste, such as by establishing infrastructure to enable food donations.

Table 40. Detailed Descriptions of Food Waste Prevention Policies

Prevention Policy	Description	Category
Education Campaigns Promoting Behavior Changes	Education campaigns focused on behavior changes can target a variety of audiences and focus on various aspects of food waste prevention. These aspects include proper food preparation, portion sizes, food reuse, ordering flexibility in restaurants, food purchasing, food storage, food safety, and meal planning. The campaigns may be done through various media outlets, including (but not limited to) mailings, face-to-face training, email, and social media.	Skills
Education Campaigns Promoting the Importance of Food Waste Prevention in Terms of Environmental, Social, and Economic Impacts	Education campaigns addressing what the issue of food waste and why it is an environmental, economic, and social concern. These programs can focus on moral issues of wasting food and the potential to save money by preventing food waste. The campaigns may be done through various media outlets, including mailings, face-to-face training, email, and social media.	Values
Food Redistribution/Donation Policies (for edible retail and commercial food)	Polices can encourage the redistribution of edible food for human consumption. Recovery policies may include tax incentives for donators, limited liability regulations for donators (such as the federal Bill Emerson Good Samaritan Food Donation Act), programs to facilitate the connection between donators and those in need, or may facilitate logistics of collection and transport.	Logistics
Promote Food Redistribution to Animal Feed	Polices can encourage the redistribution of wasted food for animal consumption. These policies can facilitate diversion of wasted food from retail and consumer sectors to animal feed, such as foods that were refused due to packaging errors or blemishes. Programs may facilitate the connection between donators and those in need, provide tax incentives to donators, or may facilitate logistics of collection and transport. Furthermore, at the household level, education can encourage people to feed excess food to pets instead of disposing it.	Logistics
Incentivize Food Waste Prevention	Policies can be enacted to incentivize prevention, such as rewarding companies that are able to significantly prevent food waste. Incentives can be financial, such as tax credits, or require higher costs for waste disposal (thus encouraging reduction).	Logistics
Increase Research and Development	Policies to support research and development can contribute to innovations which may reduce food wastage. These include improved packaging that extends shelf life, improvements in food storage, or better tracking systems for stock management. Policies may include funding for research organizations or tax incentives.	Logistics
Improve Food Packaging	Polices can encourage reconfiguration of product packaging to prevent waste, such as packaging to extend shelf life or protect products. Policies may include financial incentives to businesses using preferred packaging.	Logistics
Improve Food Date Labeling	Polices to eliminate ambiguous food labeling include well-defined, clear, scientifically-sound date labeling systems for food.	Logistics
Change Waste Collection System Design	Polices to change the design of municipal waste collection systems can help prevent food waste. These include volume based systems for trash or reduced number of days that trash is collected.	Logistics
Change Treatment of Collected Wastes	Policies can reduce food waste by stipulating how it is to be treated. An example is legislation to ban landfilling of organics. Fiscal incentives, such as taxes, fees, or subsidies, can also dictate treatment methods.	Logistics
Mandate Targets for Prevention	Policies to mandate reporting of food waste statistics and achievement of specific prevention goals.	Logistics

3.2 Selecting the Best Policy Approach

There is no one-size-fits-all solution to food waste; policy measures to address it should be custom tailored for each individual region, integrate community needs (Williams and Kelly 2003), and involve a package of several measures addressing Values, Skills and Logistics. Holistic approaches which integrate education, financial aspects, and logistical improvements across food and waste systems are ideal. It is unclear which combination of mechanisms to prevent food waste is most effective because evaluations of food waste prevention policies are scarce. Moreover, it is difficult to demonstrate a consistent, direct link between specific policy mechanisms and measured waste prevention results (Cox et al. 2010). Further complicating food waste prevention is the fact that many food waste prevention initiatives are still in their early stages, so comprehensive data are not yet available (BioIntelligence Service et al. 2010). Rather than struggle with the lack of existing data and concrete conclusions regarding the best policy means to prevent food waste, it is suggested that new, well-planned intervention campaigns be initiated, but with mandates for proper monitoring and evaluation. These data can serve as critical resources for designing future waste prevention programs. Prevention initiatives targeting food loss (losses at production, post-harvest, and processing stages of the food supply chain) should parallel food waste prevention campaigns to address the issue from multiple angles.

4. Challenges to Food Waste Prevention

There are regulatory, social, and political obstacles to enacting food waste prevention policies. The waste management framework (described in chapter five) can help ensure that these issues are addressed prior to the implementation of policies, and assists with improving systems over time (Table 41). It encourages engagement, communication, and interchanges of information across diverse stakeholders in food and waste systems to effectively prevent waste. Here key challenges associated with food waste prevention policies and ways in which the framework addresses them are discussed.

Table 41. Using the Framework to Address Challenges with Food Waste Prevention

Challenges	How Framework Addresses It
Poor Public Participation	Clearly defined target population; Carefully planned initiatives and integration of stakeholder concerns
Perceived High Costs	Thorough assessment of economic costs of policies can be used to encourage behavioral changes
Inconsistent Definitions	Stipulations for definitions of key terms
Lack of Complete Data	Stipulations for continual data collection, analysis and well defined performance metrics
Lack of Effective Indicators to Evaluate System Performance	Guidance on indicator development which cross disciplines
Little Stakeholder Engagement	Engagement of a range of stakeholders for policy development
Uncertainty Regarding Policy Performance	Consistent, thorough data collection and indicator monitoring will provide future guidance on policies that are effective and those that are not

4.1 Poor Public Participation

Many source separation programs for traditional recyclables have not succeeded because of insufficient or un-sustained citizen participation (Poulsen 2013). To address this, the waste framework encourages stakeholder engagement, especially from the public and community groups, early in the policy planning process. This communication can indicate how consumers feel about food waste and which policies will resonate with them (UNEP 2014). Assessing motivations for wasting food and openness to a prevention program may be an appropriate means to determine which efforts will be effective. A survey in Greece indicated that people had positive attitudes towards reducing food waste, they were concerned about food waste, and they understood good habits for reducing waste. Researchers also found that 405 of participants misunderstood food date labels (Abeliotis et al. 2014). A survey of over 1,000 Americans found that consumer respondents were aware of food waste issues and that they were knowledgeable about how to reduce food waste (Neff 2014). These findings suggest that many people are aware of the problem, and understand some prevention measures. Therefore, it may be beneficial to target consumers with messages that treat them as already-knowledgeable and engaged. Messages about food safety, freshness, and the meaning of food date labels to reduce disposal of still-safe foods may be effective. Neff (2014) suggests that budget focused messages are useful because consumers may be likely to waste less if they realized how much money can be saved.

Studying recycling participation indicates ways to make food waste prevention plans successful, and shows behavioral changes which may positively impact food waste initiatives. A major driver of participation in recycling programs comes from social pressure and growing environmental awareness. As Ackerman (1997) noted, many people may be concerned about the threat of global warming or the destruction of rainforests (agreeably larger environmental problems than waste issues), but individuals cannot have an immediate effect on these problems, especially in a way which is perceptible to others. Recycling and food waste prevention provide reasons for individuals to feel they are improving the environment. Therefore, if food waste prevention is to become widespread, it is important to highlight the social responsibility that individuals have to protecting the environment, as well as to show that the food waste prevention is socially and economically valuable. Recycling involves a high level of visibility (Ackerman 1997), so the behavior is partially governed by social pressure and norms. Waste prevention, unlike recycling, is done out of sight of others so and has no descriptive social norm (Cox et al. 2010). This is a challenge to food waste prevention; other means to encourage food waste prevention behavior, such as campaigns focused on environmental and social stewardship, and economic benefits are needed. Generally, the options to engage people in waste prevention behavior are not drastically different than the approaches used to increase recycling participation (Cox et al. 2010), but specifics will differ. Addressing these social aspects are important aspects of the framework.

4.2 Perceived High Costs

A barrier to implementing a waste prevention program is that participants may perceive prevention as costly, particularly for retailers and businesses that consider food waste to be inevitable and necessary for profit. It may appear beneficial in terms of labor, time, and money for restaurants to keep excess food in stock so that they never run short, even though this excess is often discarded (Buzby et al. 2011). Supermarkets keep shelves full even at the expense of throwing out excess food. Through stakeholder communication, managers can better understand why organizations feel food waste is inevitable, and work together to reconfigure processes to reduce discards. Businesses and consumers are more likely to actively prevent food waste if it is economically attractive (UNEP 2014). Options include financial incentives for businesses to reduce waste, such as sustainability certification programs, which may make a business more attractive for consumers. Some consumers seek products and services that are clearly identified as sustainable, even if they are more expensive (Harris 2007, Blackman and Rivera 2011). The direct economic costs of wasted food may also encourage businesses to reduce wastage if it results in increased profits. In restaurants, an approach to reduce plate waste is to offer various sized portions and allow selection of meal accompaniments (side dishes). This may appeal to health and environmentally conscious consumers, potentially resulting in increased sales.

4.3 Inconsistent Definitions

A challenge when planning and evaluating waste management systems is the lack of clear definitions. Food waste definitions are not universally agreed upon (Lebersorger and Schneider 2011), which makes studying and quantifying food waste difficult (Buzby and Hyman 2012), especially when comparing results across studies (Garrone et al. 2014). The framework encourages clear definitions of key terms and performance indicators to address this issue. If waste data are quantified consistently between programs and the same definitions exist for waste streams, waste management systems can be accurately compared. Clear terms also facilitate determination of performance changes over time.

4.4 Lack of Data

The lack of reliable data on food waste is a reoccurring obstacle. This was discussed in chapter three, and attempts were made to resolve some of the data gaps on food waste disposal in MSW. Poor data make it difficult to study the environmental impacts of food waste, develop and implement sound prevention policies, and to track progress over time. This issue is widespread throughout the waste field, and poor quality or unavailable data prohibits accurate system analyses and comparisons between programs (Chowdhury 2009). Data on waste prevention are especially scarce and/or poor (Cox et al. 2010). A major component of the waste management framework is the establishment of methods for the collection of sufficient data on a regular basis, and analyzing well-defined performance indicators. By implementing prevention campaigns with mandates for regular monitoring and evaluation, some of the existing data gaps will be resolved with time. These data can be an important resource for designing future waste prevention programs and can indicate which policy measures are the most effective. It also facilitates assessments of programs over time, and comparisons across regions.

4.5 Lack of Effective Indicators to Evaluate System Performance

The waste management framework encourages a transition away from solely using recycling or diversion rates to measure waste system performance. Shifting away from diversion based targets may encourage waste planners to incorporate prevention initiatives, as well as general sustainability concerns into waste management systems. A valuable indicator when examining waste prevention is the per capita disposal rate. Unlike a diversion rate, which may be high although waste disposal is high, a disposal indicator tells you the amount of food waste that is being disposed after prevention and diversion. A benefit of this indicator is that is leaves less room for ambiguity or obfuscation (than the recycling or diversion rate) with regards to calculations. Recycling and diversion rates have been shown to be ambiguous, poorly defined, calculated using different formulas, and inconsistent regarding what materials are included in calculations (Greene and Tonjes 2014). Therefore, it is difficult to monitor progress over time using these vague indicators, and nearly impossible to compare performance from one system to another.

The use of the per capita disposal rate allows for an assessment of waste prevention, which may (but not necessarily) be a result of implemented waste prevention strategies. However, changes in waste disposal may be a result of waste prevention policy, but it also could be a result of an economic change or other factor (Bartl 2014). Because prevention may be linked to consumption shifts or cultural changes, rather than just prevention initiatives, it is impossible to isolate the effect of specific waste prevention measures (Wilts 2012). Although measuring waste prevention is difficult and there generally is little understood about how to monitor and evaluate it (Sharp et al. 2010), it is nevertheless important. Evaluating disposal rates (in conjunction with other indicators) may be the best means to do so. Furthermore, if waste prevention is to be the priority, then it is essential to determine if it is occurring, even if the definitive cause cannot be absolutely assessed. Measuring indicators over time is a key step of the framework. Per capita rates are not affected by population changes, so they can serve as consistent measurements within a system over time, as well as across systems that differ in size and demographics.

Although the per capita disposal rate is important for system evaluations, it does not indicate overall environmental quality or sustainability of a waste system. The waste framework encourages using performance indicators that focus on key areas of system sustainability, including issues related to environmental, economic, and social concerns, such as degree of environmental education, number of people participating in program, amount of food redistributed to the needy, or stakeholder acceptance of programs. The most appropriate indicators may vary depending on local situations, waste system design, and political objectives (Greene and Tonjes 2014); the framework is effective for determining which indicators should be used.

4.6 Little Stakeholder Engagement

There are many opportunities for meaningful partnerships to prevent food waste (Finn 2014); prevention will require specific changes from all sectors (retail, commercial, consumer, institutional), and will need strong linkages and communication among stakeholders. The EU emphasized that tackling food waste involves working together with all stakeholders to better identify, measure, understand, and find solutions to food waste. All actors in the food chain need to work together to find solutions, including farmers, processors, manufacturers, retailers, and consumers, as well as technical experts, research scientists, food banks, and NGOs (European Commission 2014). The waste management framework encourages communication among these stakeholders, and incorporation of their concerns into policy.

4.7 Uncertainty Regarding Policy Performance

Because food waste management is still in its infancy, there is a strong need to carefully analyze existing programs to determine their performance. Currently it is unclear which food waste prevention mechanisms are most successful because evaluations of the effectiveness of the various policy options for food waste prevention are scarce, particularly because measurement of the policy impact is often not performed, especially at the local level. Because the waste framework emphasizes data collection and system evaluations, systems that utilize it can serve as key examples of what works and what does not work for food waste management. This will enable future programs to be more successful. Furthermore, a key aspect of the framework is the documentation of challenges faced when implementing policies. This information can be important when implementing similar policies elsewhere.

5. Food Waste Diversion

Food waste prevention policies can substantially reduce the amount of food waste disposed, making it an effective alternative to collection and treatment of wastes economically, socially, and environmentally. However, even with rigorous prevention programs, retail and consumer food waste will never be eliminated because some food waste is unavoidable (Schott et al. 2013), and redistribution of edible food to feed humans may be unfeasible due to food perishability and high transport or distribution costs (Buzby et al. 2014). Food also may not meet safety or quality requirements under food safety regulations (Salhofer et al. 2008). Furthermore, prevention activities may not broadly appeal to consumers and they may be costly

(Buzby et al. 2011). Therefore, enacting both prevention and diversion policies may be effective. However, by beginning with a food waste prevention initiative first (before a diversion program), many of the obstacles to food waste diversion may be reduced, as discussed below.

5.1 Public Investment

A hurdle to a successful food waste diversion program is that the system may require substantial investment from generators to source separate their wastes. Source separation programs for traditional recyclables have not been completely successful because of insufficient or un-sustained citizen participation, possibly due to inconvenience, lack of storage space, and resistance to change (Poulsen 2013). Source separating food waste is more difficult than recyclables as food waste decomposes, giving it an odor and unpleasant nature, complicating storage before collection. Lack of storage space for source separated food waste has also been identified as a barrier to food waste diversion. Furthermore, people may not want to participate in food waste diversion programs if it is costly. A 2013 survey of 2,000 Americans by the National Waste and Recycling Association found that 67 percent of those not currently in a food waste composting program would divert food waste if it was convenient to do so, but most (62 percent) would not support an increase in the cost of their waste and recycling services associated with adding food waste recovery (Harris Interactive 2013).

By preventing food wastes before diverting it, there will be less food waste to source separate, and less waste to store. This makes the programs easier to participate in, and reduces negative aspects of source separation. Less food waste to manage may reduce system costs, as there will fewer collection trucks and smaller treatment facilities; these savings may be passed on to generators, making them more likely to accept diversion programs.

5.2 Facility Siting and Logistical Challenges

There is often considerable community resistance to siting public infrastructure due to negative externalities experienced through changes in the local environmental and the presence of new risks (Esaiasson 2014). Siting municipal waste management facilities is often one of the most problematic steps in waste management. Food waste composting facilities, which are known to create objectionable odors, are a particular concern (Gutierrez et al. 2014). Composting facilities closer to residential areas need stricter odor control measures, such as in-vessel systems, which are more costly than open systems. Because food waste is heavy and wet in nature, it presents logistical challenges for transportation from the generator to processing facility, and may require specialized collection trucks. Another challenge identified for retailers, wholesalers, and manufacturers is that there may be limited food waste haulers and treatment facilities for food wastes (BSR 2013). Curbside collection of food waste also presents other challenges, such as noise and traffic from collection vehicles.

If food waste is minimized before implementing a diversion program, there will be less food waste that needs to be collected, managed, and treated, theoretically leading to a need for fewer collection vehicles and treatment facilities. This minimizes siting issues, as fewer sites are needed, and reduces negative impacts associated with collection. If there is less food waste to be managed, the obstacle of not having sufficient levels of waste transportation and management sites is also minimized.

5.3 Capacity

Organics processing capacity needs to significantly increase for the feasible recovery of more food waste. Currently, there are about 350 composting facilities and several AD facilities accepting food waste in the U.S. The addition of food waste to the 3,453 existing yard waste composting facilities may be a source of increased capacity (Platt et al. 2014). However, converting a yard waste composting operation to one that also accepts food waste poses considerable challenges because most of the current operations are not staffed or equipped to comply with the requirements for receiving food waste. Limited staff and budgets at municipal sites further complicate the transition from a solely yard waste facility to a mixed food and yard waste composting operation (Platt et al. 2014). Food waste reductions resulting from prevention programs will reduce the amount of food waste remaining in the disposal stream which needs to be diverted. So, less capacity is needed to treat the remaining food waste, thereby increasing the likelihood that adequate capacity will be available to treat all diverted food waste.

5.4 Finding Markets for Recovered Products

A major issue with increased capacity for food waste treatment is the potential lack of markets for large quantities of finished compost. This is complicated by the fact that because compost is a relatively heavy and low value soil amendment, it is not typically marketed far from where it is manufactured (Morris et al. 2014). Thus, markets are needed close to processing facilities. Food waste recovery is increasing in the U.S. and it is unclear whether or not there exists high end compost markets for the future amounts of compost produced (Levis et al. 2010). If the technology is to expand successfully, new markets must be developed both for high grade and low grade compost, and compost use must be encouraged over fertilizers. If food waste prevention programs are enacted before diversion programs, the amount of food waste remaining to be treated by alternative means (composting, AD) will be reduced, thus minimizing the amount of finished compost produced. By producing less finished material, the need for markets for these items are reduced, which means it is more likely that the amount of finished compost will be adequate to fill consumer needs.

5.5 Competing Interests

Implementing diversion programs before prevention may work to counter sustainability initiatives because effort will be put on diverting, rather than preventing food waste. If food waste infrastructure is sized based on maximal food waste diversion (assuming all generated food waste is diverted to composting or AD), the facilities will require this much material to operate efficiently. Therefore, there will be no incentives to prevent food waste, as it would detract from the diversion program. The installation of costly waste treatment facilities, particularly WTE, has been documented to be a barrier to waste prevention and recycling. Public and private investors have an interest in using existing infrastructure for as long as possible in order to achieve high returns from investments (Wilts 2012). A major issue with technological approaches to diverting food waste away from disposal is that they may be sized to require a large amount of feedstock to operate and they are costly to construct (Whyte and Perry 2001). Therefore, their installation may detract from food waste prevention as there are greater financial incentives to send the food waste to the treatment facility. If prevention initiatives are implemented first, treatment facilities can be properly sized (thus reducing cost), and there will not be competing interests for materials. Additionally, it will likely be inefficient to promote food waste prevention programs at the same time as aiming to increase source separated food waste for diversion because it will be difficult for participants and contradictory (Bernstad 2014).

6. Discussion

This work filled data gaps and demonstrated an effective framework to plan, implement and maintain a successful food waste prevention program. Prevention should be the primary objective of food waste management initiatives because it provides the greatest opportunities to reduce the social, environmental, and economical problems of food waste. USEPA’s WARM model showed that greatest environmental impact reductions are achieved through food waste prevention. Because food waste is such a complex, interdisciplinary issue, it requires multiple policy initiatives that can target the issue from several angles and disciplines. No single approach to waste prevention on its own is sufficient to prevent waste; rather a broad mixture of policy measures is needed (Cox et al. 2010). Policy mechanisms should be evaluated in terms of multiple sustainability indicators which encompass different aspects of the issue, including social, economic, and environmental concerns. Ideal policies will be locally specific.

Even after prevention initiatives, food will remain in the MSW stream, and these wastes may be managed through diversion programs. Some of the challenges associated with food waste diversion are due to the potential large quantities of food waste that needs to be diverted. If a comprehensive prevention plan is enacted successfully first, there will be less food waste that needs to be diverted, reducing challenges with diversion. As more food waste diversion programs are established, it is likely that innovation will drive the creation of technologies better equipped to manage and treat food waste. The unique challenges associated with food waste, such as its high weight and moisture content, have already, and will continue to, lead to innovations in equipment design and operation. With better technologies, it is likely that diversion systems will become more efficient and affordable.

7. Conclusion

Increasingly citizens, scientists, businesses, institutions, and policy makers are realizing that the current food system is unsustainable and changes are required if the world will be able to support a population of over nine billion by 2050. Reducing food waste will become an increasingly important strategy to help feed this growing human population (Godfray et al. 2010, Buzby et al. 2014). This will involve changing the ways food is produced and working to reduce food wastage. Wastage of food is a widespread phenomenon globally and it is likely that food waste generation will grow if not curbed by reduction policies.

Food waste prevention has not yet become main stream in the U.S. or abroad. Waste prevention in general has frequently been ignored in waste management, as signaled by states that define waste goals in terms of recycling or diversion, rather than using indicators that capture prevention success. Understanding the implications of food waste and adjusting attitudes and behaviors towards food in order to prevent wastage is an urgent priority. Special credit should be given for achievement at preventing waste. However, the most aggressive and effective food waste prevention initiatives will still leave considerable amounts of waste food as MSW, requiring effective waste management technologies and polices to manage these wastes sustainably. Successful programs need to be planned carefully, with consideration for diverse factors, including regulatory requirements, financial needs, environmental impacts, and social implications. Use of the holistic, interdisciplinary framework (described in chapter five) assists in developing comprehensive and effective policies for waste management.

This dissertation highlighted that food waste is a complex issue, involving numerous diverse actors across the globalized food chain. It is an issue that demands attention, research, and action. The research integrated multiple disciplines, including sociology, psychology, engineering, management, and statistics, to provide a multi-disciplinary examination of food waste in the U.S. This dissertation deepened the understanding of food waste, particularly with regards to the quantities of food waste disposed and the environmental impacts of alternative treatments. It also outlined an effective framework for waste management, and key policy mechanisms for reducing the amount of food waste that is disposed. It is proposed that food waste strategies be carefully analyzed in terms of environmental, social, and economic implications and multiple approaches be adopted simultaneously to combat food wastage from retail, consumer, and commercial sectors. The inclusion of sound food waste research into policy making is necessary for moving forward sustainably.

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Appendix A: Chapter 3 Supplementary Tables

Appendix A, Table 1. Websites Listing Waste Characterization Studies

Name	Author	URL
List of MSW Characterization Studies	USEPA	http://www.epa.gov/epawaste/conserve/tools/recmeas/msw_st_rpt.htm
Other MSW Composition Studies	MN Pollution Control Agency	http://www.pca.state.mn.us/index.php/waste/waste-and-cleanup/waste-management/solid-waste/integrated-solid-waste-management/minnesota-msw-composition-study/other-msw-composition-studies.html?nav=0

Appendix A, Table 2. Excluded Studies

Study Name	State	Scale	Sub-State Region [a]	Author, Year	Reason for Exclusion
Kalskag Solid Waste Characterization Study Report	AK	City, Village	Kalskag and Lower Kalsag Villages, Upper and Lower Kalskag Cities	Lung, 2006	Performed at businesses, schools, offices, and residential establishments
Detailed Characterization of Commercial Self-Haul and Drop Box Waste	CA	State	NA	Cascadia Consulting Group, 2006	Included drop off wastes only
San Francisco Waste Characterization Study	CA	County	San Francisco	ESA, 2006	Included many visually sorted samples
Waste Disposal and Diversion Findings for Selected Industry Groups	CA	State	NA	Cascadia Consulting Group, 2006	Focused on industry groups
City of San Jose C&D Characterization Study	CA	City	San Jose	Cascadia Consulting Group, 2008	Focused on C&D
City of San Jose Waste Characterization Study	CA	City	San Jose	Cascadia Consulting Group, 2008	No confidence interval
California 2006 Characterization and Quantification of Residuals from Materials Recovery Facilities	CA	State	NA	R.W. Beck and Cascadia Consulting Group, 2006	Focused on MRF residuals
California 2006 Detailed Characterization of Commercial Self-Haul and Drop-Box Waste	CA	State	NA	Cascadia Consulting Group, 2006	Focused on self-haul and drop-box waste
California 2006 Detailed Characterization of C&D Wastes	CA	State	NA	Cascadia Consulting Group, 2006	Focused on C&D
California 2006 Waste Disposal and Diversion Findings for Selected Industry Groups	CA	State	NA	Cascadia Consulting Group, 2006	Focused on industry groups
Waste Stream Study for the City of Fort Collins	CO	City	Fort Collins	Sloan Vazquez LLC and Clements	Combined data from Larimer County waste sort and Boulder

Study Name	State	Scale	Sub-State Region [a]	Author, Year	Reason for Exclusion
				Environmental, 2012	County waste sort
District of Columbia's Department of Public Works Residential Waste Sort	DC	City	DC	District of Columbia Dept. of Public Works, 1997	Did not include food waste
2011 Solid Waste Characterization Study for the District of Columbia	DC	City	Washington, DC	ARCADIS/Malcolm Pirnie, 2011	No confidence interval
Project Final Report for the Delaware Solid Waste Authority Waste Characterization Study	DE	State	NA	SCS Engineers, 1997	No confidence intervals for aggregate results
Pinnellas County Waste Composition Study	FL	County	Pinellas County	Pinellas County Utilities et al., 2002	Based on modeling
Georgia Statewide C&D Characterization Study	GA	State	NA	R.W. Beck, 2010	Focused on C&D
Hawaii County Waste Characterization	HI	County	Hawaii County	Cascadia Consulting Group, 2001	No confidence interval
University of Idaho Waste Characterization Study	IL	Facility	University of Idaho	Nagawiecki, 2009	Performed at the university level
2007 Massachusetts C&D Industry Study	MA	State	NA	DSM Environmental Services Inc., 2008	Focused on C&D
Waste Characterization Studies at MSW Combustors-Summary	MA	Facility	MSW Combustors	MA Dept. of Environmental Protection, 2012	No confidence interval or detailed methods
Montgomery County 2005 Waste Characterization Study	MD	County	Montgomery County	R.W. Beck, 2005	Full report unavailable
Maine: Residential Waste Characterization Study (2011)	ME	State	NA	Criner and Blackmer, 2011	No confidence interval
City of Red Wing Solid Waste Composition Study: Solid Waste Boiler Facility 2009	MN	City	Red Wing City	SWDI, 2009	Did not include food waste (only organics)
MN Construction, Demolition, and Industrial Waste Study	MN	State	NA	Foth, 2007	Focused on C&D
Newport Resource Recovery Facility Waste Characterization Study	MN	Facility	Newport MN Resource Recovery Facility	SAIC, 2012	Methods were not detailed enough; performed at facility level
Perham Resource Recovery Facility Waste Characterization Study	MN	Facility	Perham Resource Recovery Facility	Stericycle, 2012	No food waste category (only organics); performed at facility level
Covanta Mennepin Resource Company Waste Characterization Study 2007	MN	Facility	Covanta Hennepin	Covanta, 2007	No food waste category (only organics); performed at facility level
Covanta Hennepin Resource	MN	Facility	Covanta	Covanta, 2012	No food waste

Study Name	State	Scale	Sub-State Region [a]	Author, Year	Reason for Exclusion
Company Waste Characterization Study 2012			Hennepin		category (only organics); performed at facility level
Hennepin County Waste Composition Study - Final Report	MN	County	Hennepin County	SAIC, 2011	Significant portion of county waste is not treated at county facility
The 2008 Missouri Waste Composition Study	MO	State	NA	Midwest, Assistance Program, 2009	Looked at all solid waste, not just MSW
The 2006-2007 Missouri Municipal Solid Waste Composition Study	MO	State	NA	Midwest Assistance Program, 2007	No confidence interval
The 1997 Missouri Waste Composition Study	MO	State	NA	Midwest Assistance Program, 1997	No confidence interval
Mecklenburg County Waste Composition Study: Summary of Results	NC	County	Mecklenburg County	SCS Engineers, 2012	Targeted specific industry groups (schools, offices and college)
Orange County, NC, Waste Composition Study 2010	NC	County	Orange County	Orange County, 2010	No details on methods
Orange County, NC, Waste Composition Study 2005	NC	County	Orange County	Orange County, 2005	No details on methods
State of Ohio Waste Characterization Study	OH	State	NA	Engineering Solutions and Design Inc., 2004	No confidence interval
Rhode Island Solid Waste Composition Study Final Report	RI	State	NA	RISWMC, 1990	No details on methods; did not include food waste category
2008 Tennessee Waste Characterization Study	TN	State	NA	Tennessee State University, 2008	Sample size was not provided
Vermont Waste Composition Study 2002	VT	State	NA	DSM Environmental Services Inc., 2002	No confidence interval
Waste Composition Analysis	WA	State	NA	Matrix Group, 1988	No confidence interval
King County Monitoring Program: 2007 Waste Characterization Study	WA	County	Kings County	Cascadia Consulting Group, 2008	No confidence interval
2011 King County Waste Characterization and Customer Survey Report	WA	County	Kings County	Cascadia Consulting Group, 2012	No confidence interval
Seattle Public Utilities, 2010 Residential Recycling Stream Composition Study	WA	City	Seattle	Cascadia Consulting Group, 2010	Did not provide disposal tonnages
City of Fitchburg 2012 Waste Sort Results Report	WI	City	Fitchburg	Fitchburg Public Works, 2012	Looked at 40 households directly

Study Name	State	Scale	Sub-State Region [a]	Author, Year	Reason for Exclusion
City of Fitchburg 2009 Waste Sort Results Report	WI	City	Fitchburg	Fitchburg Public Works, 2012	Looked at 40 households directly

[a] NA = not applicable;

Appendix A, Table 3. Included Studies

ID	Study Name	Year	State	Scale	Author, Year Published
32	City of Phoenix Characterization of Waste from Single-Family Residences	2003	AZ	City	Cascadia Consulting Group, 2003
4	Alameda County Waste Characterization Study 2000	2000	CA	County	R.W. Beck, 2009
5	Alameda County Waste Characterization Study 1995	1995	CA	County	R.W. Beck, 2009
22	CA Statewide Waste Characterization Study	2008	CA	State	Cascadia Consulting Group, 2009
23	CA Statewide Waste Characterization Study	2004	CA	State	Cascadia Consulting Group, 2004
23	CA Statewide Waste Characterization Study	1999	CA	State	Cascadia Consulting Group, 2009
33	City of Los Angeles Waste Characterization and Quantification	2001	CA	City	Cascadia Consulting Group, 2002
36	City of San Diego Waste Characterization Study	2000	CA	City	Cascadia Consulting Group, 2000
52	Palo Alto Waste Composition Study	2005	CA	City	Cascadia Consulting Group, 2006
53	Sonoma County Waste Management Agency Waste Characterization Study	2007	CA	County	Cascadia Consulting Group, 2007
54	City of Sunnyvale Waste Characterization Report	2010	CA	City	Cascadia Consulting Group, 2010
3	Alameda County Waste Characterization Study 2008	2008	CA	County	R.W. Beck, 2009
77	City of San Diego Waste Characterization Study 2012-2013	2013	CA	City	Cascadia, 2014
2	2010 Boulder County Waste Composition Study	2010	CO	County	Cascadia Consulting Group, 2010
55	Larimer County Two-Season Waste Composition Study	2006	CO	County	MSW Consultants, 2007
57	Connecticut State-wide Solid Waste Composition and Characterization Study, Final Report	2009	CT	State	DSM Environmental Services Inc., 2010
6	Delaware Solid Waste Authority Statewide Waste Characterization	2007	DE	State	Cascadia Consulting Group, 2007
56	Alachua County Waste Composition Study	2009	FL	County	Townsend et al., 2010
17	Georgia Statewide Waste Characterization Study	2004	GA	Region	R.W. Beck, 2005
25	City and County of Honolulu Waste Characterization Study	2006	HI	County	R.W. Beck, 2007
11	Iowa Solid Waste Characterization 2006	2005	IA	State	R.W. Beck, 2006
12	Iowa Solid Waste Characterization 1998	1998	IA	State	R.W. Beck, 1998
48	2011 Iowa Statewide Waste Characterization Study	2011	IA	State	MSW Consultants, 2011
49	Cedar Rapids/Linn County Solid Waste Agency	2010	IA	Region	R.W. Beck, 2010

ID	Study Name	Year	State	Scale	Author, Year Published
	Waste Characterization Study				
14	Illinois Commodity/Waste Generation and Characterization Study	2008	IL	State	CDM, 2009
27	Chicago Dept. of Environment Waste Characterization Study	2009	IL	City	CDM, 2010
47	MSW Characterization Study for Indiana	2009	IN	County	Abramowitz and Sun, 2012
29	Montgomery County, MD Waste Composition Sampling and Analysis Study 2008-2009	2009	MD	County	SCS Engineers, 2009
58	Montgomery County Waste Composition Study Summary of Results	2013	MD	County	SCS, 2013
21	Minnesota MSW Composition Study 1999-2000	1999	MN	State	R.W. Beck, 2000
50	MN 2013 Statewide Waste Characterization	2013	MN	State	Burns and McDonnell, 2013
10	Final Report State of Nebraska Waste Characterization Study	2008	NE	State	Engineering, Solutions and Design, Inc., 2009
80	2004-2005 NYC Residential and Street Basket Waste Characterization Study	2004	NY	City	R.W. Beck, 2005
81	2005 Onondaga County Waste Quantification and Characterization Study	2005	NY	County	Dvirka and Bartilucci Consulting Engineers, 2006
15	1998 Oregon Solid Waste Characterization and Composition	1998	OR	State	Oregon Dept. of Environmental Quality, 1998
16	2002 Oregon Solid Waste Characterization and Composition	2002	OR	State	Oregon Dept. of Environmental Quality, 2002
74	2005 Oregon Waste Characterization Study	2005	OR	State	Oregon Dept. of Environmental Quality, 2014
75	2009/2010 Oregon Waste Characterization Study	2010	OR	State	Oregon Dept. of Environmental Quality, 2014
9	PA Statewide Municipal Waste Composition Study	2002	PA	State	R.W. Beck, 2003
82	2010 Philadelphia Waste Characterization Study	2010	PA	City	MSW Consultants, 2010
	City of Philadelphia Municipal Waste Composition Analysis Final Report	2000	PA	City	CDM, 2000
51	Sioux Falls Regional Sanitary Landfill Waste Characterization Study	2006	SD	City	R.W. Beck, 2007
44	Vermont Waste Composition Study	2012	VT	State	DSM Environmental Services Inc., 2013
20	2009 Washington Statewide Waste Characterization Study	2010	WA	State	Cascadia Consulting Group, 2010
46	Snohomish County Waste Composition Study	2009	WA	County	Green Solutions, 2009
59	Thurston County 2009 Waste Composition Study	2009	WA	County	Green Solutions, 2009
60	Thurston County 2004 Waste Composition Study	2004	WA	County	Green Solutions, 2005
61	Thurston County 1999 Waste Composition Study	1999	WA	County	Green Solutions, 2000
62	Seattle Public Utilities 2010 Residential Waste Stream Composition Study FINAL Report	2010	WA	City	Cascadia Consulting Group, 2011
63	Seattle Public Utilities 2006 Residential Waste	2006	WA	City	Cascadia Consulting

ID	Study Name	Year	State	Scale	Author, Year Published
	Stream Composition Study FINAL Report				Group, 2007
64	Seattle Public Utilities 2002 Residential Waste Stream Composition Study FINAL Report	2002	WA	City	Cascadia Consulting Group, 2003
65	Seattle Public Utilities 1998-99 Residential Waste Stream Composition Study FINAL Report	1999	WA	City	Cascadia Consulting Group, 2000
66	Seattle Public Utilities 1994-1995 Residential Waste Stream Composition Study FINAL Report	1995	WA	City	Cascadia Consulting Group, 1995
67	Seattle Public Utilities 1990 Residential Waste Stream Composition Study FINAL Report	1990	WA	City	Matrix Group, 1991
68	Seattle Public Utilities 1988/1989 Residential Waste Stream Composition Study FINAL Report	1989	WA	City	Matrix Group, 1989
69	2012 Commercial and Self-Haul Waste Streams Composition Study Final Report	2012	WA	City	Cascadia Consulting Group, 2012
70	2012 Commercial and Self-Haul Waste Streams Composition Study Final Report	2008	WA	City	Cascadia Consulting Group, 2008
71	2004 Commercial and Self-Haul Waste Streams Composition Study Final Report	2004	WA	City	Cascadia Consulting Group, 2005
72	2000 Commercial and Self-Haul Waste Streams Composition Study Final Report	2000	WA	City	Cascadia Consulting Group, 2002
73	1996 Commercial and Self-Haul Waste Streams Composition Study Final Report	1996	WA	City	Cascadia Consulting Group, 1997
18	Wisconsin Statewide Waste Characterization Study	2002	WI	State	Cascadia Consulting Group, 2003
19	Wisconsin Statewide Waste Characterization Study	2009	WI	State	MSW Consultants, 2010

Appendix A, Table 4. Studies Included in Total MSW Group

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Proportion Food Waste (90% bounds)	Season [c]
2	CO	County	Boulder County	2010	C	80	0.14 (.070, .210)	F, Su
4	CA	County	Alameda County	2000	W	2075	0.12 (.113, .127)	W, S, Su, F
5	CA	County	Alameda County	1995	W	1799	0.11 (.102, .108)	W, S, Su, F
9	PA	State	NA	2002	E	1185	0.12 (.113, .131)	W, S, Su, F
10	NE	State	NA	2008	C	624	0.17 (.158, .182)	W, S, Su, F
11	IA	State	NA	2005	C	300	0.11 (.093, .122)	F
12	IA	State	NA	1998	C	424	0.11 (.096, .118)	F, S
14	IL	State	NA	2008	C	315	0.14 (.135, .145)	F
15	OR	State	NA	1998	W	1367	0.18 (.172, .193)	W, S, Su, F
16	OR	State	NA	2002	W	844	0.16 (.147, .165)	W, S, Su, F
17.01	GA	Region	Atlantic Region	2004	E	100	0.12 (.107, 139)	W, S, Su, F
17.02	GA	Region	Central Savannah River	2004	E	50	0.14 (.118, .163)	W, S, Su, F
17.03	GA	Region	Chattahoochee Flint	2004	E	40	0.12 (.095,.136)	W, S, Su, F

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Proportion Food Waste (90% bounds)	Season [c]
17.04	GA	Region	Coastal Georgia	2004	E	31	0.11 (.089, .136)	W, S, Su, F
17.05	GA	Region	Coosa Valley	2004	E	40	0.13 (.105, .166)	W, S, Su, F
17.06	GA	Region	Georgia Mountains	2004	E	40	0.07 (.060, .083)	W, S, Su, F
17.07	GA	Region	Heart of Georgia-Altamaha	2004	E	41	0.12 (.099, .150)	W, S, Su, F
17.08	GA	Region	Lower Chattahoochee	2004	E	40	0.14 (.108, .170)	W, S, Su, F
17.09	GA	Region	Northeast Georgia	2004	E	30	0.13 (.106, .156)	W, S, Su, F
17.011	GA	Region	South Georgia	2004	E	40	0.14 (.108, .170)	W, S, Su, F
17.012	GA	Region	Southeast Georgia	2004	E	39	0.14 (.111, .164)	W, S, Su, F
17.013	GA	Region	McIntosh Trail	2004	E	30	0.07 (.059, .119)	W, S, Su, F
17.014	GA	Region	Middle Georgia	2004	E	37	0.11 (.079, .151)	W, S, Su, F
17.015	GA	Region	Southwest Georgia	2004	E	37	0.11 (.079, .151)	W, S, Su, F
20	WA	State	NA	2010	W	530	0.18 (.159, .201)	W, S, Su, F
21	MN	State	NA	1999	C	390	0.12 (.113, .137)	F
22	CA	State	NA	2008	W	751	0.16 (.141, .179)	W, S, SU, F
23	CA	State	NA	2004	W	550	0.15 (.124, .176)	W, S, Su, F
24	CA	State	NA	1999	W	1682	0.16 (.100, .220)	W, Su
25	HI	County	Honolulu County	2006	W	100	0.13 (.111, .149)	Su
29	MD	County	Montgomery County	2009	E	239	0.20 (.172, .220)	S, F
46	WA	County	Snohomish County	2009	W	201	0.15 (.098, .195)	W, S, Su, F
47.01	IN	County	Bartholomew County	2009	C	20	0.14 (.118, .164)	Su, F
47.02	IN	County	Adams County	2009	C	21	0.13 (.114, .156)	S, Su
47.03	IN	County	Davies County	2009	C	20	0.14 (.119, .158)	Su
48	IA	State	NA	2011	C	460	0.13 (.119, .148)	S, Su
49	IA	Region	Cedar Rapids/Linn County	2010	C	55	0.15 (.107, .192)	F
50	MN	State	NA	2013	C	178	0.18 (.152, .203)	S, Su
51	SD	City	Sioux Falls	2006	C	50	0.16 (.129, .202)	S
56	FL	County	Alachua County	2009	E	39	0.14 (.137, .152)	W, S
57	CT	State	NA	2009	E	258	0.14 (.128, .152)	F, W
58	MD	County	Montgomery County	2013	E	300	0.23 (.210, .246)	W, S, Su, F

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Proportion Food Waste (90% bounds)	Season [c]
59	WA	County	Thurston County	2009	W	259	0.17 (.104, .230)	W, S, Su, F
60	WA	County	Thurston County	2004	W	240	0.14 (.076, .196)	W, S, Su, F
61	WA	County	Thurston County	1999	W	268	0.15 (.101, .207)	W, S, Su, F
74	OR	State	NA	2005	W	713	0.16 (.147, .168)	W, S, Su, F
75	OR	State	NA	2010	W	950	0.17 (.161, .179)	W, S, Su, F
3	CA	County	Alameda County	2008	W	2320	0.19 (.182, .194)	W, S, Su, F
81	NY	County	Onondaga County	2005	E	49	0.15 (.116, .177)	F

[a] NA = not applicable;
[b] Region Key: C = Central U.S.; W = West U.S.; E = East U.S.
[c] Season Key: W = Winter; S = Spring; Su = Summer; F = Fall

Appendix A, Table 5. Studies Included in Sector Group

ID	State	Sector [a]	Scale	Sub-State Region [b]	Year	Region [c]	# of Samples	Proportion Food Waste	Season [d]
2.2	CO	C	County	Boulder County	2010	C	36	0.149	F, Su
4.2	CA	C	County	Alameda County	2000	W	477	0.162	W, S, Su, F
5.2	CA	C	County	Alameda County	1995	W	512	0.149	W, S, Su, F
6.2	DE	C	State	NA	2007	E	192	0.136	W, S, Su, F
9.2	PA	C	State	NA	2002	E	555	0.118	W, S, Su, F
10.1	NE	C	State	NA	2008	C	231	0.163	W, S, Su, F
11.2	IA	C	State	NA	2005	C	128	0.103	F
12.2	IA	C	State	NA	1998	C	161	0.102	F, S
14.2	IL	C	State	NA	2008	C	146	0.122	F
16.4	OR	C	State	NA	2002	W	87	0.227	W, S, Su, F
18.2	WI	C	State	NA	2002	C	166	0.132	F, Su
19.2	WI	C	State	NA	2009	C	114	0.114	F, Su
20.2	WA	C	State	NA	2010	W	164	0.220	W, S, Su, F
21.2	MN	C	State	NA	1999	C	141	0.118	F
22.2	CA	C	State	NA	2008	W	250	0.154	W, S, Su, F
23.2	CA	C	State	NA	2004	W	200	0.188	W, S, Su, F
25.2	HI	C	County	Honolulu County	2006	W	42	0.124	Su
27.2	IL	C	City	Chicago	2009	C	166	0.212	Su, F, W
44.2	VT	C	State	NA	2012	E	60	0.112	Su, F
46.2	WA	C	County	Snohomish	2009	W	96	0.131	W, S, Su, F

ID	State	Sector [a]	Scale	Sub-State Region [b]	Year	Region [c]	# of Samples	Proportion Food Waste	Season [d]
				County					
48.2	IA	C	State	NA	2011	C	247	0.131	S, Su
49.2	IA	C	Region	Cedar Rapids/Linn County	2010	C	29	0.136	F
51.2	SD	C	City	Sioux Falls	2006	C	21	0.177	S
52.2	CA	C	City	Palo Alto	2005	W	31	0.345	F
53.2	CA	C	County	Sonoma County	2007	W	115	0.266	W, S, Su, F
54.2	CA	C	City	Sunnyvale	2010	W	21	0.225	S, Su
55.2	CO	C	County	Larimer County	2006	C	39	0.159	Su, F
57.2	CT	C	State	NA	2009	E	160	0.132	F, W
59.2	WA	C	County	Thurston County	2009	W	61	0.187	W, S, Su, F
60.2	WA	C	County	Thurston County	2004	W	53	0.165	W, S, Su, F
61.2	WA	C	County	Thurston County	1999	W	60	0.150	W, S, Su, F
69.1	WA	C	City	Seattle	2012	W	259	0.298	W, S, Su, F
70.1	WA	C	City	Seattle	2008	W	271	0.316	W, S, Su, F
71.1	WA	C	City	Seattle	2004	W	270	0.299	W, S, Su, F
72.1	WA	C	City	Seattle	2000	W	347	0.250	W, S, Su, F
73.1	WA	C	City	Seattle	1996	W	348	0.222	W, S, Su, F
75.2	OR	C	State	NA	2010	W	82	0.243	W, S, Su, F
3.2	CA	C	County	Alameda County	2008	W	568	0.261	W, S, Su, F
2.1	CO	R	County	Boulder County	2010	C	34	0.131	F, Su
6.1	DE	R	State	NA	2007	E	192	0.118	W, S, Su, F
9.1	PA	R	State	NA	2002	E	630	0.122	W, S, Su, F
10.2	NE	R	State	NA	2008	C	284	0.177	W, S, Su, F
11.1	IA	R	State	NA	2005	C	86	0.112	F
12.1	IA	R	State	NA	1998	C	113	0.108	F, S
14.1	IL	R	State	NA	2008	C	169	0.146	F
16.3	OR	R	State	NA	2002	W	142	0.246	W, S, Su, F
18.1	WI	R	State	NA	2002	C	116	0.134	F, Su
19.1	WI	R	State	NA	2009	C	86	0.175	F, Su
20.1	WA	R	State	NA	2010	W	148	0.227	W, S, Su, F
21.1	MN	R	State	NA	1999	C	106	0.120	F
22.1	CA	R	State	NA	2008	W	251	0.254	W, S, Su, F
23.1	CA	R	State	NA	2004	W	150	0.173	W, S, Su, F
24.1	CA	R	State	NA	1999	W	228	0.200	W, Su
25.1	HI	R	County	Honolulu County	2006	W	44	0.137	Su

ID	State	Sector [a]	Scale	Sub-State Region [b]	Year	Region [c]	# of Samples	Proportion Food Waste	Season [d]
36.1	CA	R	City	San Diego	2000	W	354	0.139	F, W, S
44.1	VT	R	State	NA	2012	E	40	0.167	Su, F
48.1	IA	R	State	NA	2011	C	213	0.136	S, Su
49.1	IA	R	Region	Cedar Rapids/Linn County	2010	C	24	0.187	F
51.1	SD	R	City	Sioux Falls	2006	C	15	0.124	S
53.1	CA	R	County	Sonoma County	2007	W	115	0.355	W, S, Su, F
55.1	CO	R	County	Larimer County	2006	C	31	0.174	Su, F
57.1	CT	R	State	NA	2009	E	98	0.137	F, W
62.1	WA	R	City	Seattle	2010	W	361	0.290	W, S, Su, F
63.1	WA	R	City	Seattle	2006	W	356	0.334	W, S, Su, F
64.1	WA	R	City	Seattle	2002	W	309	0.329	W, S, Su, F
65.1	WA	R	City	Seattle	1999	W	360	0.267	W, S, Su, F
66.1	WA	R	City	Seattle	1995	W	368	0.201	W, S, Su, F
67.1	WA	R	City	Seattle	1990	W	114	0.183	W, S, Su, F
68.1	WA	R	City	Seattle	1989	W	212	0.163	W, S, Su, F
75.1	OR	R	State	NA	2010	W	163	0.279	W, S, Su, F
77.1	CA	R	City	San Diego	2013	W	451	0.179	F, W, S
79.1	PA	R	City	Philadelphia	2010	E	235	0.104	F, S
80.1	NY	R	City	New York	2004	E	200	0.159	S
82.1	PA	R	City	Philadelphia	2000	E	258	0.106	W, S, Su, F

[a] Sector Key = R = residential; C = commercial; [b] NA = not applicable;
[c] Region Key: C = Central U.S.; W = West U.S.; E = East U.S.; [d] Season Key: W = Winter; S = Spring; Su = Summer; F = Fall

Appendix A, Table 6. Studies Included in Geographical Classification Group

ID	State	Sector [a]	Rural /Urban	Scale	Sub-State Region [b]	Year	Region [c]	# of Samples	Proportion Food Waste	Season [d]
14.4	IL	T	Ru	State	NA	2008	C	63	0.143	F
16.2	OR	T	Ru	State	NA	2002	W	487	0.158	W, S, Su, F
21.4	MN	T	Ru	State	NA	1999	C	140	0.145	F
21.1.2	MN	R	Ru	State	NA	1999	C	27	0.132	F
21.2.2	MN	C	Ru	State	NA	1999	C	28	0.154	F
47.02	IN	T	Ru	County	Adams County	2009	C	21	0.135	S, Su
47.03	IN	T	Ru	County	Davies County	2009	C	20	0.138	Su
14.3	IL	T	Ur	State	NA	2008	C	252	0.131	F
16.1	OR	T	Ur	State	NA	2002	W	349	0.153	W, S, Su, F
21.3	MN	T	Ur	State	NA	1999	C	240	0.110	F
21.1.1	MN	R	Ur	State	NA	1999	C	80	0.115	F
21.2.1	MN	C	Ur	State	NA	1999	C	121	0.108	F
27.2	IL	C	Ur	City	Chicago	2009	C	166	0.212	Su, F, W
27.1.1	IL	R	Ur	City	Chicago	2009	C	214	0.199	Su, F, W
27.1.2	IL	R	Ur	City	Chicago	2009	C	20	0.149	Su, F, W
32.1.1	AZ	R	Ur	City	Phoenix	2003	C	283	0.168	W, Su
33.1.1	CA	R	Ur	City	Los Angeles	2001	W	80	0.269	W, Su
36.1	CA	R	Ur	City	San Diego	2000	W	354	0.139	F, W, S

[a] Sector Key: R = residential; C = commercial
[b] NA = not applicable; [c] Region Key: C = Central U.S.; W = West U.S.; E = East U.S.
[d] Season Key: W = Winter; S = Spring; Su = Summer; F = Fall

Appendix A, Table 7. Studies Included in Per Capita Food Waste Disposal Rate Group

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Rate [c]	Season [d]
2	CO	County	Boulder County	2010	C	80	0.579	F, Su
3	CA	County	Alameda County	2008	W	2320	0.825	W, S, Su, F
4	CA	County	Alameda County	2000	W	2075	0.701	W, S, Su, F
5	CA	County	Alameda County	1995	W	1799	0.647	W, S, Su, F
9	PA	State	NA	2002	E	1185	0.500	W, S, Su, F
10	NE	State	NA	2008	C	624	0.700	W, S, Su, F
11	IA	State	NA	2005	C	300	0.424	F
12	IA	State	NA	1998	C	424	0.445	F, S
14	IL	State	NA	2008	C	315	0.818	F
15	OR	State	NA	1998	W	1367	0.734	W, S, Su, F
16	OR	State	NA	2002	W	844	0.667	W, S, Su, F

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Rate [c]	Season [d]
17.01	GA	Region	Atlantic Region	2004	E	100	0.617	W, S, Su, F
17.02	GA	Region	Central Savannah River	2004	E	50	0.369	W, S, Su, F
17.03	GA	Region	Chattahoochee Flint	2004	E	40	0.391	W, S, Su, F
17.04	GA	Region	Coastal Georgia	2004	E	31	0.481	W, S, Su, F
17.05	GA	Region	Coosa Valley	2004	E	40	0.373	W, S, Su, F
17.06	GA	Region	Georgia Mountains	2004	E	40	0.306	W, S, Su, F
17.07	GA	Region	Heart of Georgia-Altamaha	2004	E	41	0.325	W, S, Su, F
17.08	GA	Region	Lower Chattahoochee	2004	E	40	0.438	W, S, Su, F
17.09	GA	Region	Northeast Georgia	2004	E	30	0.457	W, S, Su, F
17.011	GA	Region	South Georgia	2004	E	40	0.528	W, S, Su, F
17.012	GA	Region	Southeast Georgia	2004	E	39	0.782	W, S, Su, F
17.013	GA	Region	McIntosh Trail	2004	E	30	0.459	W, S, Su, F
17.014	GA	Region	Middle Georgia	2004	E	37	0.395	W, S, Su, F
17.015	GA	Region	Southwest Georgia	2004	E	37	0.395	W, S, Su, F
20	WA	State	NA	2010	W	530	0.742	W, S, Su, F
21	MN	State	NA	1999	C	390	0.426	F
22	CA	State	NA	2008	W	751	0.922	W, S, SU, F
23	CA	State	NA	2004	W	550	0.905	W, S, Su, F
24	CA	State	NA	1999	W	1682	0.913	W, Su
25	HI	County	Honolulu County	2006	W	100	0.719	Su
29	MD	County	Montgomery County	2009	E	239	0.723	S, F
46	WA	County	Snohomish County	2009	W	201	0.518	W, S, Su, F
47.01	IN	County	Bartholomew County	2009	C	20	0.829	Su, F
47.02	IN	County	Adams County	2009	C	21	0.207	S, Su
47.03	IN	County	Davies County	2009	C	20	0.562	Su
48	IA	State	NA	2011	C	460	0.531	S, Su
49	IA	Region	Cedar Rapids/Linn County	2010	C	55	0.583	F
50	MN	State	NA	2013	C	178	0.526	S, Su
51	SD	City	Sioux Falls	2006	C	50	0.932	S
56	FL	County	Alachua County	2009	E	39	0.489	W, S
57	CT	State	NA	2009	E	258	0.494	F, W
59	WA	County	Thurston County	2009	W	259	0.645	W, S, Su, F
60	WA	County	Thurston County	2004	W	240	0.597	W, S, Su, F
61	WA	County	Thurston County	1999	W	268	0.597	W, S, Su, F
74	OR	State	NA	2005	W	713	0.686	W, S, Su, F

ID	State	Scale	Sub-State Region [a]	Year	Region [b]	# of Samples	Rate [c]	Season [d]
75	OR	State	NA	2010	W	950	0.630	W, S, Su, F

[a] NA= not applicable
[b] Region Key: C=Central U.S.; W=West U.S.; E= East U.S.
[c] Pounds/person/day
[d] Season Key: W= Winter; S= Spring; Su= Summer; F = Fall

Appendix A, Table 8. USEPA Food Waste Estimates

Estimate Year	Percent of Food Waste in Disposed MSW	Thousands of Tons of Food Waste Disposed per year	Per Capita Disposal Rate (pounds/person/day)	Report Year [a]
1960	14.8%	12,200	0.37	1995 and newer
	14.9%	12,200	0.37	1988, 1990
1965	13.1%	12,700	0.36	1990
	12.9%	12,400	0.35	1988
1970	11.3%	12,800	0.34	1990 and newer
	11.4%	12,800	0.34	1988
1975	11.3%	13,400	0.34	1990
	11.5%	13,400	0.34	1988
1980	9.5%	13,000	0.31	1995 and newer
	9.8%	13,200	0.32	1990
	9.2%	11,900	0.29	1988
1981	9.2%	12,100	0.29	1988
1982	9.3%	12,000	0.28	1988
1983	8.9%	12,000	0.28	1988
1984	8.8%	12,200	0.28	1988
1985	9.1%	13,200	0.30	1990
1986	8.9%	12,500	0.29	1988
1988	8.5%	13,200	0.30	1990
1990	13.6%	23,800	0.52	2009/2010/2011/2012
	12.1%	20,800	0.46	1997/1999/2000/2001/ 2003/2005/2006/2007/2008
	8.1%	13,200	0.29	1996
1991	8.6%	13,660	0.30	1996
1992	12.5%	21,000	0.45	1997
	8.4%	13,560	0.29	1996
1993	8.5%	13,720	0.29	1996
1994	12.9%	21,020	0.44	1997
	8.5%	13,560	0.29	1995
	8.5%	13,390	0.28	1996
1995	13.4%	21,170	0.44	1999/2000/2001/2003
	13.5%	21,330	0.44	1998
	13.6%	21,230	0.44	1997
	8.9%	13,450	0.28	1996
1996	14.0%	21,330	0.44	1998
	14.0%	21,380	0.44	1997
1997	15.0%	24,040	0.49	1999
	13.6%	21,330	0.44	1998

Estimate Year	Percent of Food Waste in Disposed MSW	Thousands of Tons of Food Waste Disposed per year	Per Capita Disposal Rate (pounds/person/day)	Report Year [a]
1998	15.1%	24,330	0.49	1999/2000
1999	14.8%	24,610	0.49	1999/2000/2001
2000	17.3%	30,020	0.58	2011/2012
	16.8%	29,130	0.57	2009/2010
	15.4%	26,130	0.51	2007/2008
	15.6%	26,430	0.51	2006
	15.6%	25,800	0.50	2003/2005
	15.4%	25,220	0.49	2000/2001
2001	16.2%	26,250	0.50	2003
	15.8%	25,470	0.49	2001
2002	16.1%	27,180	0.52	2006
	16.1%	26,540	0.51	2003
2003	16.6%	27,760	0.52	2008
	16.6%	27,430	0.52	2005
	16.4%	26,800	0.51	2003
2004	16.7%	28,750	0.54	2007
	17.0%	29,070	0.54	2006
	16.7%	28,410	0.53	2005
2005	18.5%	32,240	0.60	2011/2012
	18.1%	31,300	0.58	2009/2010
	15.4%	29,530	0.55	2007/2008
	17.6%	29,790	0.55	2006
	17.1%	28,540	0.53	2005
2006	17.6%	30,360	0.56	2007
	18.0%	30,570	0.56	2006
2007	19.1%	32,750	0.60	2011
	18.7%	31,800	0.58	2009/2010
	18.1%	30,840	0.56	2007/2008
2008	21.3%	34,910	0.63	2012
	19.5%	32,540	0.58	2009/2010
	18.6%	30,990	0.56	2008
2009	21.3%	34,910	0.61	2011
	20.8%	33,440	0.60	2009/2010
2010	21.0%	34,770	0.62	2011/2012
	20.7%	33,790	0.60	2010
2011	21.4%	35,040	0.62	2012
	21.3%	34,910	0.62	2011
2012	21.1%	34,690	0.62	2012

[a] These data are based on review of the following report years: 1988, 1989, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2003, 2005, 2006, 2007, 2008, 2009, 2010, 2011, and 2012

Appendix A, Table 9. Tonnage Food Waste Disposed in MSW Stream and Per Capita Rates

ID	State	Scale	Sub-State Region [a]	Year	Food Waste %	Waste Shed Disposal (tons)	Food Waste Disposal (tons)	Population	Disposal Rate (Pounds/Person /Day)
2	CO	County	Boulder County	2010	14.1	220,817	31,135	294,567	0.579
3	CA	County	Alameda County	2008	18.7	1,187,108	221,989	1,474,368	0.825
4	CA	County	Alameda County	2000	11.9	1,552,683	184,769	1,443,741	0.701
5	CA	County	Alameda County	1995	10.5	1,514,450	159,017	1,346,548	0.647
9	PA	State	NA	2002	12.0	9,369,082	1,124,290	12,331,031	0.500
10	NE	State	NA	2008	17.1	1,342,000	229,482	1,796,378	0.700
11	IA	State	NA	2005	10.6	2,163,054	229,283	2,966,334	0.424
12	IA	State	NA	1998	10.7	2,203,848	235,811	2,902,872	0.445
14	IL	State	NA	2008	13.9	13,697,526	1,903,956	12,747,038	0.818
15	OR	State	NA	1998	18.2	2,468,309	448,985	3,352,449	0.734
16	OR	State	NA	2002	15.6	2,743,561	427,996	3,513,424	0.668
17.01	GA	Region	Atlantic Region	2004	12.2	3,164,338	386,049	3,429,379	0.617
17.02	GA	Region	Central Savannah River	2004	14.0	205,143	28,720	426,482	0.369
17.03	GA	Region	Chattahooc-hee Flint	2004	11.5	182,543	20,992	294,076	0.391
17.04	GA	Region	Coastal Georgia	2004	11.1	454,429	50,442	574,283	0.481
17.05	GA	Region	Coosa Valley	2004	13.4	282,651	37,875	556,207	0.373
17.06	GA	Region	Georgia Mountains	2004	7.1	403,828	28,672	513,054	0.306
17.07	GA	Region	Heart of Georgia-Altamaha	2004	12.3	134,998	16,605	279,589	0.325
17.08	GA	Region	Lower Chattahooc-hee	2004	13.8	148,023	20,427	255,792	0.438
17.09	GA	Region	Northeast Georgia	2004	13.0	309,996	40,300	483,435	0.457
17.011	GA	Region	South Georgia	2004	13.7	150,770	20,656	214,520	0.528
17.012	GA	Region	Southeast Georgia	2004	13.6	166,145	22,596	158,287	0.782
17.013	GA	Region	McIntosh Trail	2004	7.4	160,592	11,884	141,773	0.459
17.014	GA	Region	Middle Georgia	2004	11.3	405,302	45,799	635,199	0.395
17.015	GA	Region	Southwest	2004	11.3	405,302	45,799	635,199	0.395

ID	State	Scale	Sub-State Region [a]	Year	Food Waste %	Waste Shed Disposal (tons)	Food Waste Disposal (tons)	Population	Disposal Rate (Pounds/Person /Day)
2	CO	County	Boulder County	2010	14.1	220,817	31,135	294,567	0.579
3	CA	County	Alameda County	2008	18.7	1,187,108	221,989	1,474,368	0.825
			Georgia						
20	WA	State	NA	2010	18.3	4,978,496	911,065	6,724,543	0.742
21	MN	State	NA	1999	12.4	2,997,450	371,684	4,775,508	0.427
22	CA	State	NA	2008	15.5	39,722,818	6,157,037	36,604,337	0.922
23	CA	State	NA	2004	14.6	40,235,328	5,874,358	35,574,576	0.905
24	CA	State	NA	1999	15.7	35,535,453	5,579,066	33,499,204	0.913
25	HI	County	Honolulu County	2006	12.7	940,187	119,404	909,683	0.719
29	MD	County	Montgome-ry County	2009	19.6	654,471	128,276	971,600	0.723
46	WA	County	Snohomish County	2009	14.6	460,700	67,400	713,335	0.518
47.01	IN	County	Bartholome w County	2009	14.1	81,402	11,502	76,063	0.829
47.02	IN	County	Adams County	2009	13.5	9,579	1,291	34,256	0.207
47.03	IN	County	Davies County	2009	13.8	22,692	3,141	30,620	0.562
48	IA	State	NA	2011	13.3	2,233,506	297,056	3,064,102	0.531
49	IA	Region	Cedar Rapids/Linn County	2010	14.6	188,077	27,459	257,940	0.583
50	MN	State	NA	2013	17.8	2,922,045	520,124	5,420,380	0.526
51	SD	City	Sioux Falls	2006	16.4	153,759	25,217	148,244	0.932
56	FL	County	Alachua County	2009	14.1	154,234	21,747	243,574	0.489
57	CT	State	NA	2009	13.5	2,379,687	321,258	3,561,807	0.494
59	WA	County	Thurston County	2009	16.7	176,578	29,542	250,979	0.645
60	WA	County	Thurston County	2004	13.6	178,800	24,370	223,535	0.597
61	WA	County	Thurston County	1999	15.5	144,500	22,340	204,873	0.598
74	OR	State	NA	2005	15.7	2,874,644	452,182	3,613,202	0.686
75	OR	State	NA	2010	17.0	2,596,340	441,118	3,837,208	0.630

[a] NA= not applicable