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WATER AND ENERGY COSTS OF LANDFILLED FOOD WASTE

by

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ABSTRACT

Energy and water are consumed or contaminated during both the production and disposal of wasted food. To date, evaluations of water and energy resources associated with food waste have considered only resources used in food production. To allow for the full characterization of food waste within a Food Energy Water (FEW) nexus framework, this study addresses a fundamental knowledge gap related to the energy and water impacts of food waste after disposal. Fluxes of water and energy related to disposal of wasted food in landfills within the state of Florida were characterized. It is estimated that each metric ton (Mg) of landfilled food waste produces 18.1 kWh of energy, while the energy needed for collection, leachate transport, and treatment totals 126.5 kWh/Mg. These values equate to a net energy cost of 108.4 kWh/Mg, which is 110 Million kWh annually in Florida. It was observed that the water footprint of landfilled food waste is related to the assimilation of contaminated effluent and ranges from 2.5 to 58.5 m^3 per metric ton of landfilled food waste, depending on the constituent of interest. Up to 58 Million m^3 of water may be required annually to assimilate contamination related to landfilled food waste in Florida.

We assessed the sensitivity of 14 variables used to estimate energy and water impacts and found that impacts are sensitive to the proportion of landfills collecting and utilizing landfill gas, concentration of constituents in leachate, and volume of effluent. Future research should be focused to improving the characterization of these influential parameters, and to similar FEW analysis of other food waste management technologies, such as composting or anaerobic digestion. Better understanding of water and energy impacts of food waste could inform societal decision making regarding investment in FEW-efficient waste management technologies.

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CHAPTER ONE: INTRODUCTION

Food waste is a national and global challenge, with consequences to food security and environmental sustainability. Food waste is a significant energy and nutrient rich waste stream. Energy and water are consumed and produce contaminants both during the production of wasted food and in the management process. Carbon and other nutrients generated from food waste are potentially problematic and contaminants require energy intensive processes to treat along with large volume of water to assimilate.

When food is wasted, there is little thought to the resources lost or the environmental impacts of waste management. Energy is a typical metric when quantifying the resources wasted when conducting a life-cycle assessment. Another factor that needs to be considered is the water associated with food production, processing, consumption, and disposal. The food-energy-water (FEW) nexus recognizes that these resources are interconnected. Within the FEW nexus, energy and water costs of food production are conceptually understood and as economic commodities relatively well-described. Food production requires resources which include land, energy, and water and is identified as one of the earth's most energy-intensive industries (Chameides et al., 1994) ranking third behind steel production and petroleum refining (Heichel,1976). To better understand the potential impacts of wasted food within the FEW nexus, estimates of water and energy related to food waste disposal and management of food waste nutrients are needed.

The objective of this study is to identify locations, directions, and magnitudes of energy and water fluxes related to landfilled food waste in the State of Florida. This analysis is the first to

quantify impacts of food waste management within the FEW nexus, and is visually depicted in Figure 1. This thesis focuses on landfilling since this is the primary management option for handling food waste in the United States. Therefore, the fluxes related to landfilled food waste will be quantified, but also a conceptual framework will be presented that can be used to evaluate the water and energy impacts of alternative management options. This nexus study will provide a better understanding of landfilling as a food waste management option in the context of the FEW nexus. This conceptual framework can also provide a scientific approach to quantify the water and energy for other food waste management techniques such as anaerobic digestion, composting, and incineration. The outcomes of this framework are aimed at reducing the quantities of wasted food and providing recommendations on the most efficient management options for food waste.
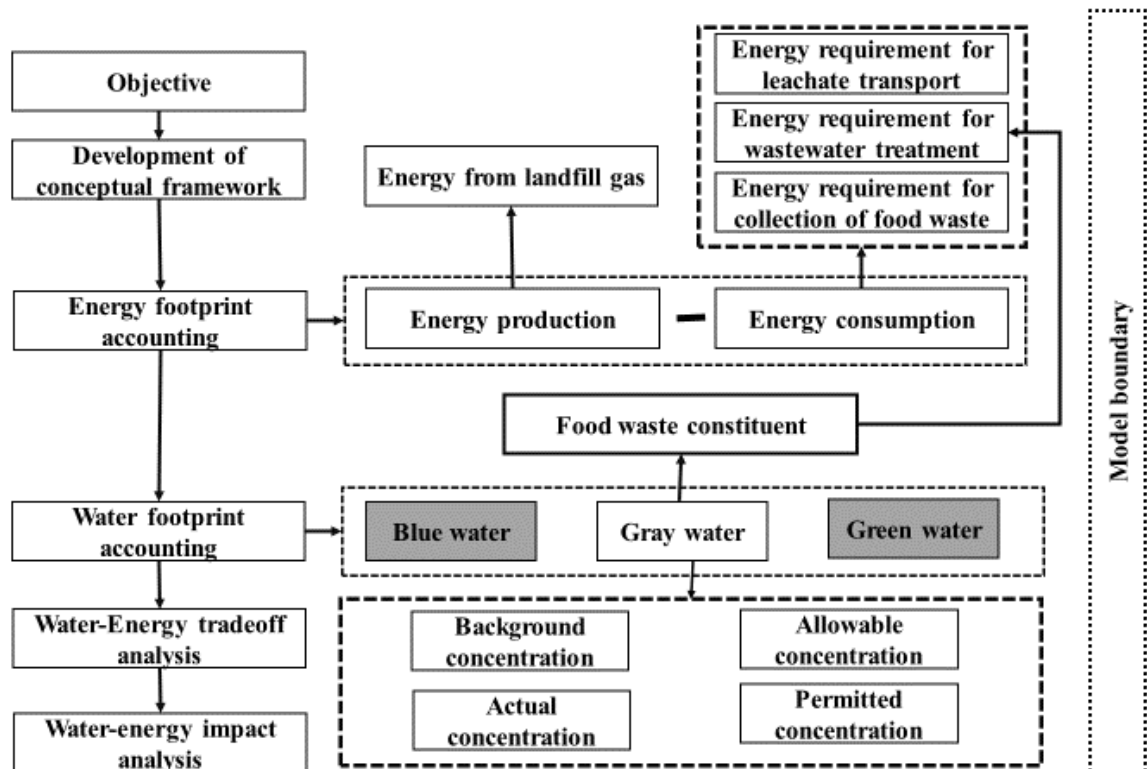
Figure 1: Research framework

CHAPTER TWO: LITERATURE REVIEW

Food waste is a subset of organic waste, which includes manure, yard debris, food processing and post consumer wastes, and agricultural wastes. Definitions of food waste presented by different organizations around the globe vary considerably based on materials included, management approaches, and means of production. (Thyberg and Tonjes, 2016). A composite of these organizations and their respective definitions is in Table 1. Notably, wastes generated during pre- and post-consumer phases are not consistently delineated within various definitions. For instance, Food and Agriculture Organization (FAO) and the European Commission definitions include both food losses and food waste from multiple phases of the value chain, while others define food waste solely within the post-consumer phase. Food loss is the amount of food which could potentially be used for consumption but which is not eaten. Whereas food waste involves the amount disregarded and not consumed by humans which refers to spoil or throw away before disposal. (Thyberg and Tonjes, 2016).

Table 1 Definitions of food waste

Organization	Definition
United Nations Food and Agriculture Organization	Food lost or wasted in the production chain leading to "edible products going to human consumption"
European Commission	Food (including inedible parts) lost from the food supply chain, not including food diverted to material uses such as bio-based products, animal feed, or sent for redistribution
United States Environmental Protection Agency	Uneaten food and food preparation wastes from residences, commercial and institutional establishments
US Department of Agriculture	A subset of food losses; occurs when an item still edible at the time of disposal is not consumed
World Resources Institute	Food fit for human consumption that is discarded—either before or after it spoils; either the result of negligence or a conscious decision to throw food away

Several studies have estimated the percentage of food that is wasted. For example, Silvennoinen et al.,2015 found that around 20% of food served is wasted from the Finnish food service system in the process of preparation and handling. Betz et al.,2014 estimated that storage, preparation, and serving losses, combined with plate waste in Switzerland, totaled around 18% of food grown. In the United States, approximately 31% of food grown was wasted in 2010 (Buzby et al., 2014). This study also found that 61% of food waste occurs in the consumption phase, 17% is generated in the production phase, and the remainder is lost during handling, storage, processing, packaging, distribution, and marketing. This total food waste in the United States represents a loss of 1,520 Kcal/per capita/per day out of an available 3,976 Kcal/per capita/per day grown globally (WRI, 2013; Buzby et al., 2014) The variability in reported food waste estimates may reflect the multiple definitions of food waste, or, possibly the true variability in behavior among locations. For example, it is estimated that 56% of the total global food waste is generated in developed

countries (defined as North America, Europe, Oceania and industrialized parts of Asia covering China, Japan and Korea), with the remaining 44% generated in the developing world (WRI, 2013).

The United States spends about one billion dollars per year in disposal of wasted food (USEPA, 2014). Wasted food is one of the largest components of the waste stream by weight in the United States, comprising over 14.5% of the total municipal solid waste (MSW) generated (by product category and by material volume) in American households. The primary mechanism for managing food waste in the Unites States is landfill disposal. Less than 3% of food waste is recovered annually through composting (USEPA, 2014). In 2013, around 2.1% of generated food waste was processed by anaerobic digestion (EREF, 2015). An unknown quantity of food waste is disposed of in garbage disposal systems which enter the sewer system for treatment at a wastewater treatment facility. The remainder of this waste was combusted at waste-to-energy facilities. The decomposition of food and other organic waste in landfills and anaerobic digesters produces methane, a greenhouse gas 21 times more potent to the environment relative to carbon dioxide. Uncontrolled landfill gas is the third largest human-related source of methane in the United States, accounting for 34% of all methane emissions (USEPA, 2014). To improve food security and conserve resources, the United States Environmental Protection Agency and United States Department of Agriculture have established a national goal to halve food waste by 2030.

Food production requires resources which include land, energy, and water and is identified as one of the earth's most energy-intensive industries (Chameides et al., 1994) ranking third behind steel production and petroleum refining (Heichel,1976). In the U.S., energy used in food

production varies from 10% to 17% of total energy production (Steinhart,1974; Hirst,1974; Heller and Keoleian, 2000; Pimentel,2008). For example, the energy production cost for corn grown in the US is approximately 4,200 kWh/Mg (Pimentel and Patzek, 2005). In Florida, the average energy requirement for agricultural production varies between 5,000 to 25,000 kWh/Mg (Fluck, 1979). It is estimated that domestic energy used for food production is increasing (Khan et al.,2009). For example, in the U.S. energy for food production grew by a factor of six between 1997 and 2002 (Canning et al., 2010). The main sources of energy for food production in the U.S. are petroleum and natural gas (Pimentel et al., 2008).

Water required to produce food has been quantified by numerous researchers (Durning and Brough, 1991; Beckett and Oltjen,1993; Mekonnen and Hoeksrta, 2010; Kreith, 1991; Aldrich et al., 1978; Pimental, 2001). On a global scale nearly 35% of the annual world water budget is required for agricultural food production (Chen and Chen, 2013). It is estimated that 67% of the global freshwater withdrawals are used for agricultural irrigation (Doll and Siebert, 2002; Foley et al., 2005,). The water footprint is a tool used to quantify the water used to grow, process, produce, and dispose of food (Hoekstra and Hung, 2002; Hoekstra, 2003; Chapagain and Hoekstra, 2007; Hoekstra et al., 2009). A total water footprint is conceptualized as the sum of three components: blue (e.g., surface and groundwater withdrawals), green (e.g., soil moisture), and grey (e.g., water used to assimilate contaminants) (Hoekstra et al., 2009). For example, the water footprint of rice production is estimated as 1,325 m^3/Mg broken down by 48% green, 44% blue, and 8% grey (Chapagain and Hoekstra, 2007). In the U.S., the total water footprint for agricultural goods is estimated to be 1,192 m^3/capita/year for domestic water footprint to produce goods and services

(Hoekstra and Chapagain, 2007). The water footprint of various horticultural products varies widely: sugar crops (roughly 200 m^3/Mg), vegetables (300 m^3/Mg), roots and tubers (400 m^3/Mg), fruits (1000 m^3/Mg), cereals (1,600 m^3/Mg), oil crops (2,400 m^3/Mg) to pulses (4,000 m^3/Mg) (Mekonnen and Hoekstra, 2011). Researchers have identified the energy and water impacts associated with the production of wasted food. For example, around 2% of total energy generated in the United States is consumed in production of wasted food (Cuellar & Webber, 2010) and 225 m^3 to 3,500m^3 of water is consumed in producing each metric ton of wasted food, (Abhat, 2015). However, research to this point has only quantified the water and energy costs related to producing wasted food (Levis et al., 2011, Kummu et al., 2012) but have not quantified the water and energy costs incurred in the post-consumer phase.

The traditional way to manage food waste in the post-consumer phase is landfilling (Levis et al., 2011). In the United States, nearly 53.8% of municipal solid waste is discarded to a landfill. (USEPA, 2014). In the United States the recycling and recovery rate is 34.5% and thermal conversion with energy recovery is 11.7 % of municipal solid waste. (USEPA, 2011) The other ways to manage food in post-consumer phase are incineration processes, composting, and anaerobic digestion. Waste collection is the first step to move any solid waste to a waste management facility. Global statistics show that up to 95% of municipal solid waste are moved to landfills (Diamadopoulos, 1994; Kurniawan and Chan, 2006) with up to 97% of food waste being discarded to landfills (Levis et al., 2011). Methane is generated from landfills from the biodegradable part of solid waste (50-60% of volume) (Shin et al., 2005; USEPA 2012). Nearly 40% of landfill gas is composed of carbon dioxide. Municipal solid waste (MSW) landfills are the

third-largest source of human-related methane emissions in the United States, accounting for approximately 18.2 percent of these emissions in 2014 (USEPA, 2016). At the same time, methane emissions from landfills represent a lost opportunity to capture and use a significant energy resource (USEPA, 2016). Landfill gas also consists of other compounds at lower concentrations such as oxygen, nitrogen, sulfur compounds, water vapor, and non-methane organic compounds (USEPA, 2000; Shin et al., 2005). Landfill gas can be a source of energy (i.e. alternative vehicular fuel proposed by Mainmoun et al., 2016 or contribution to electricity grid, Heller et al., 2004; Hirschberg, 1999; Dones et al., 2003; Boyle, 1997). To directly utilize the landfill gas in energy production, landfill gas needs to be converted to pipeline quality gas that requires a high energy content process requiring separation of carbon dioxide and other constituents from the gas stream (Hesson, 2008; USEPA, 2000).

A major concern of landfilling food waste is the leachate generation. Leachate is the liquid generated from landfilled wastes which is affected by the waste composition and amount of infiltrating precipitation (Duggan, 2005). Landfill leachate must be treated to meet state and national regulatory standards before discharged to the environment. The main constituents of concern include biochemical oxygen demand (BOD), chemical oxygen demand (COD), total ammonia-nitrogen (TAN), organic nitrogen, phosphorous, sulfate, sodium, potassium and metals (e.g. chromium, cadmium, cobalt, copper, lead, mercury, nickel, and zinc) (Kjeldsen et al., 2002). A study by WRAP,2010 suggested that food waste contains primarily BOD, TAN, chromium, cadmium, mercury, nickel and selenium. The treatment and assimilation of landfill leachate involves both water and energy intensive processes. The concentrations of the heavy metals in

food waste vary across samples around the world as presented in Table 2. The metals are relatively low in concentration and do not contribute significantly to landfill leachate.

Table 2 Concentration of metals in food waste, based on dry mass

Source	Unit	Cd	Cr	Pb	Hg	Ni	Se
Bozym et al., 2015	mg kg^{-1}	0.5	1.0	1.0	-	1.0	-
WRAP (2010)	mg kg^{-1}	<0.4	<2	3.6	<2	<1.4	<1
Pollak et al., 2004	mg kg^{-1}	0.001-2.17	0.04-10	<0.001-0.87	0.02-0.38	<0.001 - 9.55	-
Fisgativa et al., 2016	mg kg^{-1}	0.3	0.28	18	0.3	10	-
Luo et al., 2010	mg kg^{-1}	0.26-1.17	9.66-19	73.3-134	-	7.04-10.3	-

Leachate is predominantly co-treated with domestic wastewater at a wastewater treatment plant (Abbas et al., 2009). Methods published to treat leachate include combined treatment with domestic sewage; biological processing (anaerobic and anaerobic); chemical/physical treatment (flotation, coagulation/flocculation, chemical precipitation, adsorption, ammonium stripping, chemical oxidation, ion exchange and Electrochemical treatment), and membrane filtration (microfiltration, ultrafiltration, nanofiltration and reverse osmosis) (Abbas et al., 2009 and Renou et al., 2008). These studies found that biological treatment is most effective for treating landfill leachate. During aerobic biological treatment, oxygen is supplied to oxidize organic matter and nitrogen (Abbas et al., 2009; Renou et al., 2008). Heavy metals are assumed to be treated in the biological treatment by sorption onto the biomass. (Abbas et al., 2009). The released heavy metal in leachate treatment effluent is not well described. Talaraj, 2015 found that the release of heavy metals from a typical Polish MSW landfill ranges from 0.025% to 1.685% of its original

concentration of leachate. Although this concentration of heavy metals is low, a significant volume of water is required to assimilate (Talalaj, 2015) these parameters to ambient concentrations.

There have been multiples study on the LCA for solid waste management facilities which involve energy and water quantification and interdependencies. Denison,1996 used LCA to compare landfilling incineration based on solid waste output, energy use, and pollution released to the air and water. This study suggested that based on LCA incineration was favorable over landfilling considering the energy implications. Morris,2005 reviewed the energy tradeoffs between thermal conversion and landfilling. It was reported that the energy input was low for both processes (i.e., landfilling and thermal conversion) compared to the production energy costs. Arena et al.,2004 focused on the LCA of paper disposal in landfills. This study found that a significant amount of water was required to manage this waste. Recycling of food waste was shown to have lower water requirements. Despite the attempts to use LCA to make decisions regarding solid waste management there are still knowledge gaps regarding the interdependencies of water-energy interconnection during food waste management.

Modeling of energy and water impacts is subject to uncertainty due to the combined effects of data variability, measurements, estimations, unrepresentative values, missing data, and modeling assumptions (Clavreul et al., 2012). A sensitivity analysis can describe the influence of input variables in a model (Clavreul et al., 2012). Many researchers have identified different approaches for uncertainty analysis applied to solid waste management. Huang et al., 1992 applied a grey linear programming approach in solid waste management to address model stipulations and

coefficients. An updated programming technique,grey fuzzy linear programming, addresses solid waste management planning considering these model uncertainties (Huang et al., 1993). Maqsood and Huang (2003) applied a stochastic programming model for planning purposes. Cheng et al. (2003) applied linear programming for the selection of a landfill location. Lo et al., 2004 applied a Bayesian Monte Carlo Method to evaluate treatment options in LCA for greenhouse gas generation by addressing chemical compositions of constituents and heavy metals. Finnveden and his colleagues (Finnveden et al., 2005; Morberg et al., 2005) have applied uncertainty analysis for energy generation from solid waste management. All these studies are focused on total solid waste management and not food waste. Considering food waste and its management approach, there is a need to address this uncertainty. In the interaction of energy and water when considering food waste disposal options, uncertainty analysis is required to quantify the dynamics of the system.

The goal of this study is to identify the water-energy interdependencies of food waste disposal through landfilling. This goal will be achieved by quantifying the energy and water requirements to management food waste disposed of in the state of Florida using literature values (e.g., methane generation rate and potential, Florida contributory landfills and uncertainty).

CHAPTER THREE: METHODOLOGY

3.1 Background

This study focuses on water and energy usage and trade-offs for food waste management in Florida landfills. In the post-consumer phase of food waste management, energy or water have the potential to be either consumed and/or produced. The pathways are identified for landfill management and then mapped related to the water and energy fluxes for landfills which is the primary food waste disposal mechanism in the United States and Florida. To understand the potential energy and water footprint of wasted food, the fate of food waste carbon and nutrients during treatment was considered. Carbon and nutrients entering landfills as food waste are separated into gaseous (NH_3, CO_2, CH_4, NO_x) and liquid emissions (leachate), or stored within the landfill (Figure 2). During anaerobic degradation of food waste in a landfill, complex organic materials (e.g., carbohydrates, lipids, proteins) are hydrolyzed to soluble products and then converted to methane and carbon dioxide through methanogenesis.
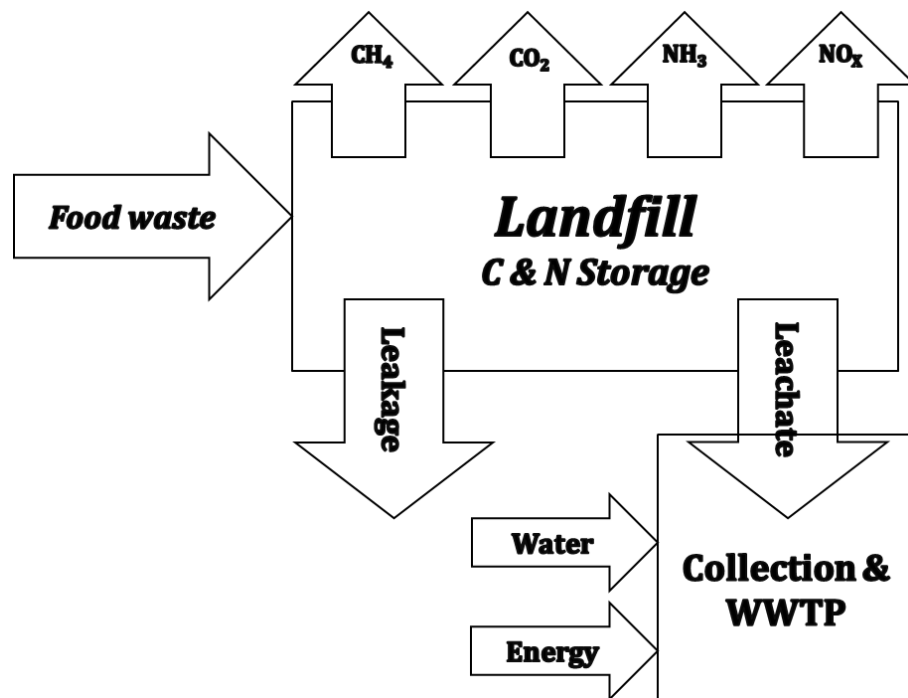
Figure 2: Fate of food waste carbon and nitrogen entering landfills

Energy can be consumed or produced if food waste is managed in a landfill as shown in Figure 3. To quantify the energy and water impacts of food waste disposal in a landfill, a conceptual map was developed of the pathway for this material from the point of disposal through the treatment and assimilation of leachate while identifying energy and water fluxes at each step. Energy production occurs through the generation of landfill gas during anaerobic degradation. The collection of waste and transportation of the solid waste to the landfill, trucking of leachate to a WWTP, and subsequent treatment consumes energy. If the landfill gas is collected and utilized, this energy production (WARM, 2015) can contribute positively to the energy grid. Lastly, treatment of landfill leachate requires both energy and/or water-intensive treatment before effluents can be discharged to the environment.

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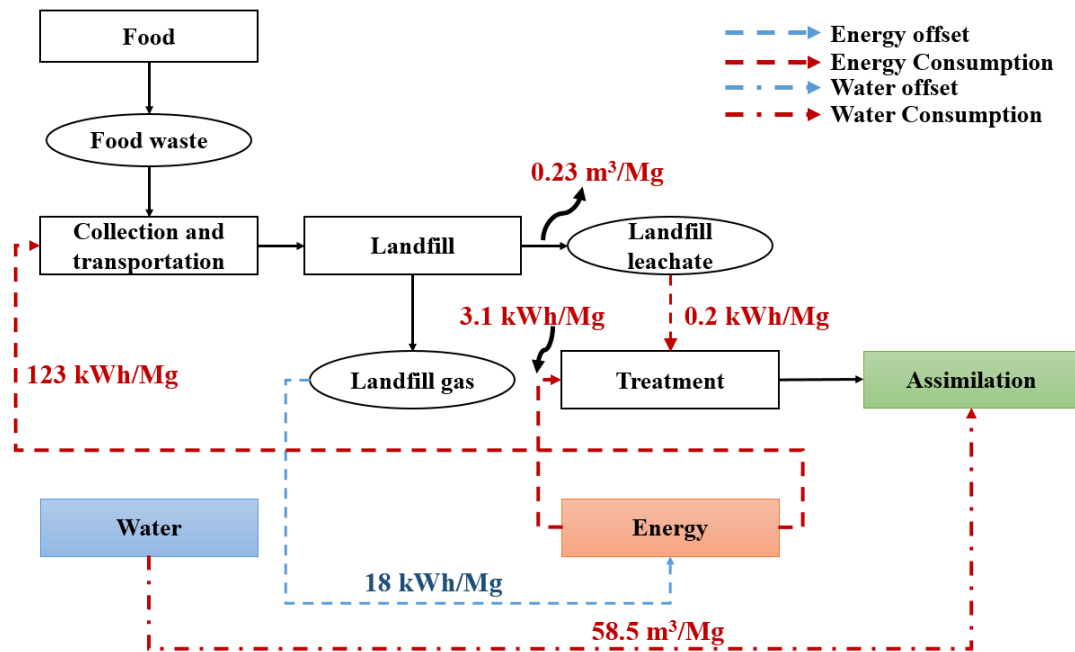
Figure 3: Landfilled food waste in the FEW Nexus

3.2 Study location

The geographical boundary for the conceptual treatment of food waste through landfilling is the state of Florida. In 2015, The State of Florida is ranked 2nd in energy generation and 3rd in energy consumption in the U.S. (USDOE, 2016). In 2015, nearly 2.3×10^{11} kWh energy is generated among which 5.3×10^9 kWh are renewables (USEIA, 2016). Total consumption for electricity was 1.21×10^{12} kWh. Freshwater withdrawal was around 2×10^{10} m^3. (USGS, 2016). On average, 39% of total available fresh water was used in agricultural and public water supply in Florida (FDEP, 2014). Florida was chosen due to the availability of local data from numerous studies focused on landfill gas production (Amini and Reinhart, 2011; Amini et al., 2011), leachate generation and management (Maimoun et al., 2013; Maimoun et al., 2016; Bolyard and Reinhart, 2016; Bolyard, 2016), and water footprints related to MSW landfills (Maimoun, 2015). Amini and

Reinhart, 2011 determined the significance of food waste in energy potential for the State of Florida and found that diversion of food waste from landfills could result in 9% decrease in methane generation, while only 1% decline in energy production potential due to the difficulty in capturing methane from the rapidly degrading, labile fractions of food waste. Leachate from various landfills across Florida was characterized by Bolyard, 2016. Maimoun, 2015 calculated the transportation and collection costs of waste management as well as the associated water footprints. Energy and water accounting is limited to food waste management by landfilling and not extended to the quantification of water and energy consumed through the construction of the landfill, transportation vehicles, and the subsequent treatment of leachate at a wastewater treatment plant (WWTP).

3.3 Energy footprint calculation

The quantity of energy (kWh) that must be expended to return contaminated discharges to ambient background concentrations. To understand net impact to the energy sector, both the energy costs and benefits of disposed food are quantified as in Eq. 1. For the quantification procedure, more than twenty parameters are introduced and used (Table 4).

$$E = E_P - (E_L + E_C + E_T) \tag{1}$$

Where,

E = net energy (kWh/Mg)

E_P = energy production (kWh/Mg)

E_L = energy for leachate treatment (kWh/Mg)

E_C = energy for food waste collection (kWh/Mg)

E_T = energy for transportation of leachate to a WWTP (kWh/Mg)

3.3.1 Energy cost for food waste collection

The energy used for food waste collection was quantified using the procedure developed by Maimoun et al., 2015. Although different assumptions were made regarding travel time, distance, and speed from the household to the landfilling site. For example, Mainmoun et al., 2015 assumed an average distance from household to household to be around 22.3 meters and the associated collection time to be approximately 8 seconds with a speed of 10 km/hr for the State of Florida. The distance the collection vehicle was required to travel to the landfill was around 19km over a 20-minute period. Fuel consumption rates for collection vehicles vary depending on operational condition; for example, collection vehicle operation frequency or whether the waste is collected in urban or rural areas. The annual energy costs (E_C, kWh/Mg) associated with collection and transportation of food waste was calculated using Eq. 2:

$$E_c = \alpha_{fc}\varepsilon_f \tag{2}$$

Where,

α_{fc} = fuel consumption rate (L/Mg)

ε_f = fossil fuel energy potential (kWh/L).

3.3.2 Energy production from landfill gas generated by food waste

Potential energy production is calculated from gas produced per metric ton (Mg) of food waste disposed in conventional Florida landfills (i.e., no leachate recirculation). We estimated the

volume of methane gas that can be collected from each unit of food waste using Eq. 3 (Amini and

Reinhart, 2011, Amini et al., 2011):

$$Q_c = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k\eta_{ij} \left(\frac{M_{FW,i}}{10}\right) L_0 e^{-kt_{zj}} \tag{3}$$

Where,

Q_c = Collected methane (m^3/Mg)

k = Methane generation constant of food waste (year^{-1}).

L_o = Methane generation potential of food waste (m^3/Mg)

j = 1/10 time increments (year)

z = time period of LFG generation from waste disposal in year i (year)

t_{zj} = age of jth section of waste M_{FW} in year z (year)

$M_{FW,i}$ = Mass of landfilled food waste disposed in year i, (Mg)

η_{ij} = Efficiency in methane gas collection (m^3/m^3)

As methane generation will vary with time since disposal, we compute mean methane collection

over the period of peak methane production, the first three years after disposal. Only 22 out of the

163 landfills in Florida currently collect landfill gas (LFG) (USEPA, 2016). After accounting for

facilities at which LFG collection is infeasible, an upper bound of 70% of Florida landfills have

the potential to contribute to the energy sector. We estimate methane energy production from food

waste (E_p, kWh/Mg) using Eq. 4,

$$E_p = \varepsilon_{CH_4} \varphi_1 \varphi_2 \varphi_3 Q_c \tag{4}$$

Where,

φ_1 = Methane to energy capacity factor (fraction, m^3/m^3);

φ_2 = Electrical efficiency (fraction, kWh/kWh);

φ_3 = Proportion of landfills collecting LFG (fraction)

ε_{CH_4} = Methane energy potential (kWh/m^3)

3.3.3 Energy cost of leachate treatment

Presumably the BOD and TAN concentrations in leachate produced strictly by food waste (food waste leachate) are different from values reported for MSW leachate (e.g. Bolyard, 2016). In order to account for these differences, we estimate leachate BOD and TAN concentrations attributed to food waste by scaling concentrations observed in MSW leachate by the proportion of carbon and nitrogen in food waste relative to the biodegradable fraction of MSW to (Eq 5-8, Figure 4).

$$BOD_{FW} = \frac{M_{FW}}{M_{MSW}} * \frac{MC_{FW}/M_{FW}}{MC_{MSW}/M_{MSW}} * BOD_{MSW} \qquad (5)$$

Where:

BOD_{FW} = BOD concentration of leachate attributed to food waste (mg/L)

BOD_{MSW} = BOD concentration of MSW leachate (mg/L)

M_{FW} = Mass of food waste (Mg)

M_{MSW} = Mass of municipal solid waste (Mg)

MC_{FW} = Mass of carbon in food waste (Mg)

MC_{MSW} = Mass of carbon in municipal solid waste (Mg)

The carbon and nitrogen content in MSW (Table 3) can be calculated from Eq.6 and Eq.8.

$$MC_{MSW} = \frac{\sum_{i=1}^{n} C_i W_i}{\sum_{i=1}^{n} W_i} \tag{6}$$

Where:

C_i = Carbon content in municipal solid waste (Mg) of biodegradable component, i

W_i = Contribution of biodegradable component, i in MSW (fraction)

$$TAN_{FW} = \frac{M_{FW}}{M_{MSW}} * \frac{MN_{FW}/M_{FW}}{MN_{MSW}/M_{MSW}} TAN_{MSW} \tag{7}$$

Where:

TAN_{FW} = TAN concentration of leachate attributed to food waste (mg/L)

TAN_{MSW} = TAN concentration of MSW leachate (mg/L)

MN_{FW} = Mass of nitrogen in food waste (Mg)

MN_{MSW} = Mass of nitrogen in municipal solid waste (Mg)

$$MN_{MSW} = \frac{\sum_{i=1}^{n} N_i W_i}{\sum_{i=1}^{n} W_i} \tag{8}$$

Where:

N_i = Nitrogen content in municipal solid waste (Mg) of biodegradable component, i

Table 3 Carbon and nitrogen in biodegradable part of MSW and their contributions in MSW

Component	Carbon content in waste stream i, C_i (Mg of Carbon / Mg of waste, i)	Nitrogen content in waste stream i, N_i (Mg of Nitrogen / Mg of waste, i)	Source	Percent of MSW, W_i (%)	Source
Biodegradable components of MSW					
Food waste	0.480	0.026	Worrell and Vesilind (2011)	14.9	USEPA, 2016
Paper	0.435	0.003		26.6	
Cardboard	0.440	0.003		26.7	
Textiles	0.550	0.046		3.2	
Rubber	0.780	0.020		3.2	
Leather	0.600	0.100		3.2	
Yard wastes	0.478	0.034		13.3	
Wood	0.495	0.002		6.2	
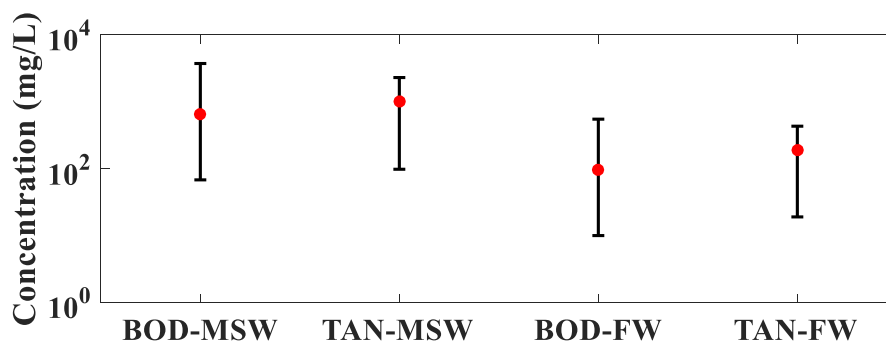
Figure 4 BOD and TAN concentrations in MSW leachate (measured) and attributable to food waste (calculated)

The amount of leachate generated by landfills varies with regional precipitation, landfilled area exposed to precipitation, and operational phase (USEPA, 2011). We estimate the generation rate of leachate ($Q_{MSW,T}$, m^3/Mg) from a MSW landfill during the phase of active landfilling (daily

cover only), estimating an active phase length of 10 years (USEPA, 2011), (Eq. 9, Camobreco et al., 1999) as:

$$Q_{MSW,T} = \frac{FP}{D\rho}t \tag{9}$$

Where:

$Q_{MSW,T}$ = Leachate generation (m^3/Mg)

F = fraction of precipitation collected as leachate (m/m);

P = annual mean precipitation (m/year)

ρ = Density of landfilled MSW (Mg/m^3)

D = Waste Depth (m)

t = time of leachate generation (years)

The energy that is utilized for treatment to return contaminated discharges to concentrations that may be permitted to discharge to the environment was calculated based on regulatory standards. The energy costs (kWh/Mg) of treating leachate can be estimated using Eq. 10.

$$E_L = Q_{MSW,T}\, E_{O_2}\left[\left(BOD_{FW} - BOD_{Reg}\right)K_1 + \left(TAN_{FW} - TAN_{Reg}\right)K_2\right] \tag{10}$$

Where:

K_1 = Oxygen requirement for BOD treatment (kg/kg)

K_2 = Oxygen requirement for TAN treatment (kg/kg)

E_{O_2} = Energy requirement for oxygen supply (kWh/Mg)

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BOD_{Reg} = Regulatory standard of BOD in receiving waterbody (mg/L)

TAN_{Reg} = Regulatory standard of TAN in receiving waterbody (mg/L)

3.3.4   Energy cost of leachate transport

It is assumed that leachate is collected from the landfill and transported to a local WWTP. Generally, leachate is carried by a heavy-duty truck to a WWTP if there is no option for discharge through the sewer system. USEPA, 2011 has estimated that the fuel consumption rate of heavy duty truck for leachate transport with a default distance of around 25km (15 miles) is 0.89 L for per Mg of leachate transported. Here, we consider a ratio of mass of MSW leachate to the mass of total MSW to convert the energy requirement in each Mg of waste. In this case, the limitation is not to consider the amount of leachate attributable to food waste. Instead, the energy considered here is the value to transport the leachate generated from municipal solid waste. Here, it is assumed that the energy required for leachate attributable to food waste is similar as the energy required for transporting municipal solid waste. The annual energy costs (E_T, kWh/Mg) associated with transportation of food waste leachate was estimated using as Eq. 11.

$$E_T = \alpha_{fT} \, \varepsilon_f \, \frac{Q_{MSW}}{\rho_{leachate}} \tag{11}$$

Where:

α_{fT}= fuel consumption rate (L/Mg of leachate)

Q_{MSW} = annual volume of leachate produced (m³/Mg) ($Q_{MSW,T}$ / t)

$\rho_{leachate}$ = Density of MSW leachate (Mg/m³)

Table 4 Energy footprint baseline parameter values

Parameter	Value	Unit	Source(s)
Energy production by landfill gas (kWh/Mg)			
Methane generation rate (k)	0.35	Year^{-1}	Machado et al. (2009), Amini and Reinhart (2011), IPCC (2006)
Methane generation potential(L_o)	300.7	m^3/Mg	Stanley and Barlaz (2011); Amini and Reinhart (2011)
Gas collection efficiency (η_{ij})	0.67	Fraction	Amini and Reinhart (2011)
Capacity factor (φ_1)	0.83	Fraction	Amini and Reinhart (2011)
Electrical efficiency (φ_2)	0.35	Fraction	Amini and Reinhart (2011)
Proportion of landfills collecting LFG (φ_3)	0.13	Fraction	USEPA (2016)
Energy potential (ε_{CH4})	10.4	kWh/m^3	Amini and Reinhart (2011)
Energy cost of food waste collection (kWh/Mg)			
Fuel consumption rate, food waste collection (α_{fc})	11.1	L/Mg	Maimoun et al. (2015)
Fossil fuel energy potential (ε_f)	11.1	kWh/L	Packer (2011)
Energy cost for leachate transport (kWh/Mg)			
Fuel consumption rate, leachate transport (α_{fT})	0.89	L/Mg	USEPA (2011), Maimoun (2015)
Density of landfilled MSW-leachate ($\rho_{leachate}$)	1120	Kg/m^3	USEPA (2016), Souza et al., (2014)
Mass of landfilled MSW (M_{MSW})	2 x10^7	Mg/year	FDEP (2014)
Energy cost for leachate treatment (kWh/Mg)			
BOD and TAN of leachate attributed to food waste			
Dry mass of carbon content of food waste (C_{FW})	48	%	Worrell and Vesilind (2011)
Dry mass of carbon content of MSW (C_{MSW})	48.6	%	Calculated, Eq. 5
BOD in leachate (BOD_{MSW})	651	mg/L	Bolyard (2016)
BOD in leachate attributed to FW (BOD_{FW})	96	mg/L	Calculated
Dry mass of Nitrogen content of food waste	2.6	%	Worrell and Vesilind (2011)
Dry mass of nitrogen content of MSW (C_{MSW})	2.1	%	Calculated, Eq. 7

Parameter	Value	Unit	Source(s)
TAN in MSW leachate (TAN_{MSW})	1020	mg/L	Bolyard (2016)
TAN in leachate attributed to FW (TAN_{FW})	191	mg/L	Calculated
Fraction of precipitation collected as leachate (F_t)	20	%	USEPA (2011), Maimoun (2015)
Annual precipitation (P)	1250	mm	NCDC (2015)
Density of landfilled MSW (ρ)	708	Kg/m^3	Worrell and Vesiland (2011)
Waste Depth (D)	15	m	Maimoun (2015)
Time for leachate generation (t)	10	years	USEPA (2011)
Regulatory standard of BOD in receiving water (BOD_{Reg})	20	mg/L	FDEP (2016a)
Regulatory standard of TAN in receiving water (TAN_{Reg})	8.75	mg/L	Bloetscher and Gokgoz (2001)
Energy requirement for O_2 supply (E_{O2})	14.2	kWh/Kg	City of Union,SC (2011)
O_2 requirement for BOD treatment (K_1)	1.5	Kg/Kg	Environmental Dynamics
O_2 requirement for TAN treatment (K_2)	4.0	Kg/Kg	International (2005)

3.4 Water footprint calculation

The grey water footprint (Hoekstra et al., 2009) of wasted food was calculated to estimate the quantity of water needed to assimilate leachate contaminents attributed to food waste. The grey water footprint can be directly quantified using a mass balance in the form of a 2-end member mixing model (Figure 5).
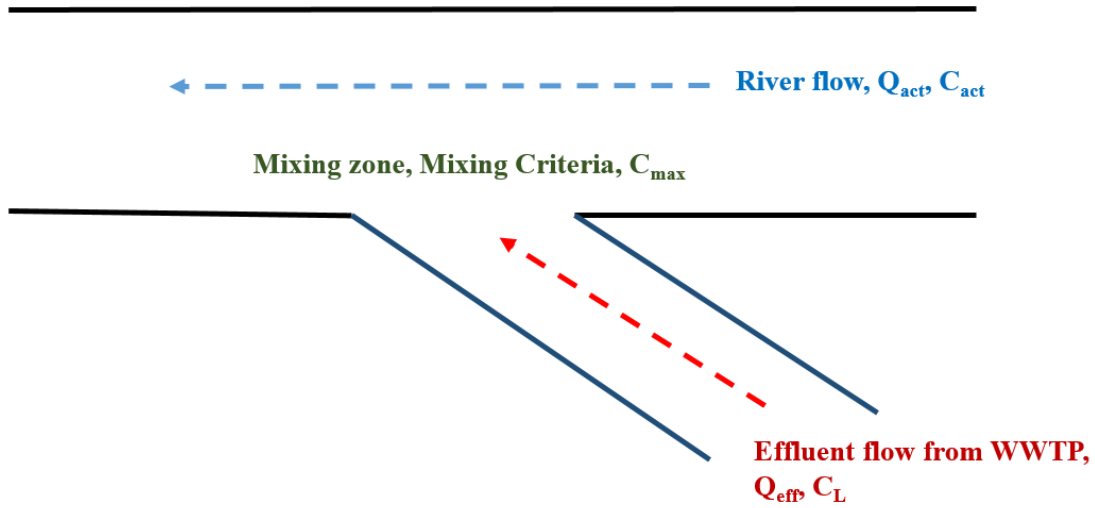
Figure 5 Illustration of 2-end member mixing model for grey water footprint calculation

Effluent from a WWTP (Q_{eff}) with some concentration of constituent *i* ($C_{L, i}$) is dischared into the free flowing water body (with concentration of $C_{act, i}$) and the mixing criteria is the maximum acceptable concentration of constrituents according to regulatory standards (C_{max}) (Figure 5). Applying 2-end member mixing model (1st end member is actual discharge; 2nd end member is effluent discharge) for the particular system, the following equations (Eq.12 and Eq.13) using conservation of mass can be found,

$$Q_{new} = Q_{act} + Q_{eff} \tag{12}$$

$$Q_{act}C_{act,i} + Q_{eff}C_{L,i} = (Q_{act} + Q_{eff})C_{max,i} \tag{13}$$

Where,

Q_{new} = Mixed discharge (m^3/s)

Q_{eff} = Effluent discharge (m^3/s)

Q_{act} = Actual flow (m^3/s)

$C_{L,i}$ = Permitted concentration of contaminant i in wastewater discharge (mg/L)

$C_{max,i}$ = Maximum acceptable concentration of contaminant i (mg/L) in receiving waterbody

$C_{act,i}$ = Actual concentration of contaminant i (mg/L) in receiving waterbody

Combining Eqs.12 and 13,

$$Q_{act} = \frac{Q_{eff}(C_{L,i} - C_{max,i})}{(C_{max,i} - C_{act,i})} \tag{14}$$

By definition, grey water is amount of water required for waste assimilation (Hoekstra et al., 2009 Hoekstra et al. ,2011); thus, according to laws of mass conservation Q_{act} is the grey water footprint. Using the leachate generation rate for each Mg of waste, Eq.14 can be rewritten for grey water footprint of each metric ton of food waste (m³/Mg, WF$_{Grey}$) as in Eq.15.

$$WF_{Grey} = \frac{Q_{MSW,T}(C_{L,i} - C_{max,i})}{(C_{max,i} - C_{act,i})} \tag{15}$$

Actual contentrations of constituents in natural water bodies may vary in time and over space. It is thus challenging to estimate a single actual concentration of constituents representative of the all receiveing waterbodies in the state of Florida. Assuming the waterbody is in compliance with regulatory standards, actual concentrations of constituents should beless than the maximum acceptable concentration and is expected to be higher than natural concentration of contaminant of the receiving water body. The natural concentration of constituents refers to representative concentrations where there is no anthropogenic impact in a water body (Hoekstra et al., 2011). We

estimate the actual concentration as a triangular fuzzy number, ranging from the background concentration to the maximum acceptable concentration and including the in the mid-point value. The water footprint model is parametrized according to permitted concentrations in wastewater effluents (nutrients and heavy metals) as outlined by the Florida Administrative Code (FAC, Chapter 62-4, FDEP, 2016a). The maximum concentration of BOD that may be discharged to receiving waters is 20 mg/L (FDEP, 2016a). As TAN concentrations are dynamic, varying with temperature and pH (Chapman, 1996), no firm standard is applied to all state waters. Rather, the determination of permitted discharge concentration is conducted by local administrative bodies and requires site-specific negotiations. It is assumed that mean concentration of TAN is 8.75 mg/L discharged to water bodies, based upon the average TAN concentration observed in secondary effluents for South Florida water bodies (Bloetscher and Gokgoz, 2001) (Table 5).

In Florida, BOD concentration is regulated such that dissolved oxygen (DO) shall not be depressed below the permitted limit (FDEP, 2016a). As DO varies with season, water column depth, and characteristics of the specific water body, there is not a single regulatory concentration for BOD applied to all Florida waterbodies, at all times. The statewide screening level concentration for BOD is 2.0 mg/L which is the 70th percentile of all data across the State of Florida from 1970 to 1987 (FDEP, 2008; FDEP, 2013). Chapman,1996 reported that unpolluted water typically has a BOD value of 2 mg/L, while Hand,2004 calculated a typical BOD value of Florida surface waters ranges from 0.50 mg/L to 3.40 mg/L with a mean value of 1.30 mg/L. Based on the available literature, we use 2.0 mg/L as our maximum acceptable concentration of BOD. We modeled a representative maximum TAN concentration for Florida waters given ranges of temperature and

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pH for Florida streams (Hand, 2004) using relations provided by FAC Chapter 62-302.500 (FDEP, 2016b). Mean TAN concentration over a 30-day period in a Florida waterbody should be equal or less than 1.4 mg/L. Maximum acceptable concentrations of nutrients and heavy metals in a water body are given by the FAC (FAC, Chapter 62-302, FDEP, 2016b) for surface waters (Table 5). We apply estimates of background concentration for BOD, TAN and metals from Hand, 2004 (Table 5).

Table 5 Water footprint baseline parameter values

Parameter	Value	Unit	Source(s)
Leachate volume ($Q_{MSW,T}$)	0.23	m^3/Mg	See Table 4; Section 3.3.4
Biochemical Oxygen Demand (BOD)			
$C_{L,BOD}$	20	mg/L	FDEP 2016(a)
$C_{max,BOD}$	2	mg/L	FDEP 2016(b)
$C_{background,BOD}$	0.5	mg/L	Hand (2004)
Total Ammonium-Nitrogen (TAN)			
$C_{L,TAN}$	8.75	mg/L	Bloetscher and Gokgoz (2001)
$C_{max,TAN}$	1.4	mg/L	FDEP 2016(b)
	1.0-1.8 (range)	mg/L	
$C_{background,TAN}$	10	µg/L	Hand (2004)
	15	µg/L	Franke el al. (2014)
Cadmium (Cd)			
$C_{L,Cd}$	0.1	mg/L	FDEP 2016(a)
$C_{max,Cd}$	8.8	µg/L	FDEP 2016(b)
$C_{background,Cd}$	0.0	mg/L	Hand (2004)
	0.001	µg/L	Franke el al. (2014)
Chromium (Cr)			
$C_{L,Cr}$	0.5	mg/L	FDEP 2016(a)
$C_{max,Cr}$	0.011	mg/L	FDEP 2016(b)
$C_{background,Cr}$	0.0	mg/L	Hand (2004)
	0.1	µg/L	Franke et al. (2014)
Lead (Pb)			
$C_{L,Pb}$	0.5	mg/L	FDEP 2016(a)
$C_{max,Pb}$	8.5	µg/L	FDEP 2016(b)
$C_{background,Pb}$	0	mg/L	Hand (2004)
	0.4	µg/L	Franke et al. (2014)
Mercury (Hg)			
$C_{L,Hg}$	1.5	µg/L	FDEP 2016(a)
$C_{max,Hg}$	0.012	µg/L	FDEP 2016(b)
$C_{background,Ni}$	0	mg/L	Hand (2004)
Nickel (Ni)			
$C_{L,Ni}$	1	mg/L	FDEP 2016(a)
$C_{max,Ni}$	8.3	µg/L	FDEP 2016(b)
$C_{background,Ni}$	0	mg/L	Hand (2004)
	0.4	µg/L	Franke et al. (2014)
Selenium (Se)			
$C_{L,Se}$	0.1	mg/L	FDEP 2016(a)
$C_{max,Se}$	5.0	µg/L	FDEP 2016(b)
$C_{background,Se}$	0	mg/L	Hand (2004)

3.5 Sensitivity analysis

Sensitivity analysis is performed based on the variability of parameters in context to water and energy footprint. We introduce thirty parameters for energy quantification and five parameters for water quantification. Among the thirty parameters, fourteen are selected for sensitivity analysis in energy quantification (Cases: A-N) as the other sixteen parameters are assumed to be constant and four of them for water sensitivity analysis.

3.6 Uncertainty analysis

In this section, tentative direction towards addressing uncertainty is approached. The following framework (Figure 6) is introduced to quantify uncertainty within the FEW nexus.
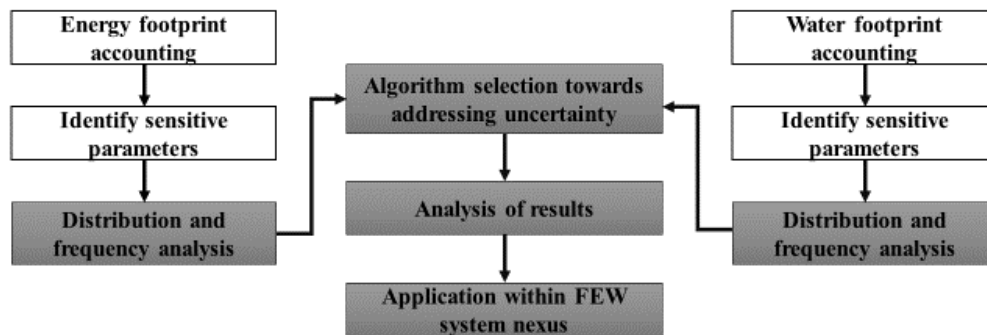
Figure 6 Uncertainty analysis framework within FEW nexus

An uncertainty range was estimated for energy and water footprint using the modeling approach in the previous sections for this case study. Monte Carlo simulation calculates the models for a specified number of times, each time using different randomly-selected values. A Monte Carlo computational method is introduced to determine uncertainty of output values for energy and water footprint using the most sensitive parameters values and from a specified range and

probability distribution for input parameters, i.e. BOD and TAN concentrations, contributory landfills proportionality and leachate generation rate. All the parameter values are adjusted to minimum and maximum values so that while applying those values in the above equations, actual data points fell within the energy and water quantification. All these parameters are assumed to be normally distributed (Figure 7).
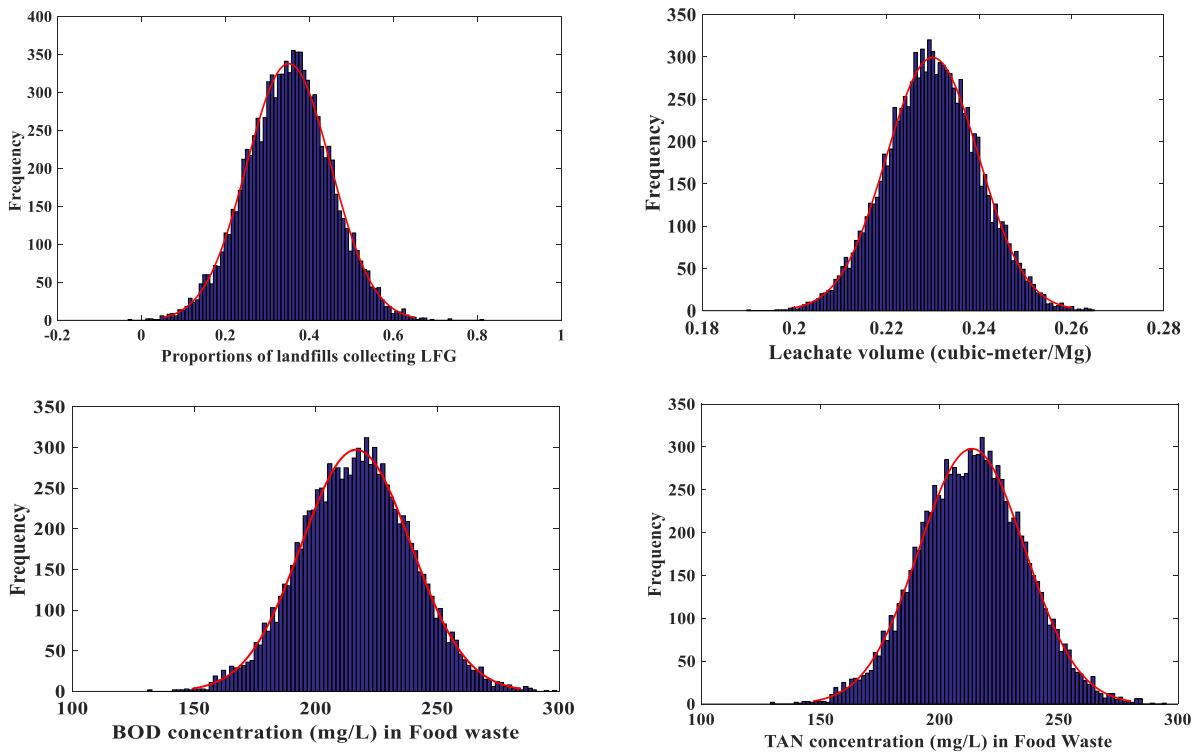

Figure 7 Normal distribution of sensitive parameters

The Monte-Carlo analysis is simulated for 10000 runs. The analysis produces 10000 combinations for water and energy footprint values.

CHAPTER FOUR: RESULTS

4.1 Energy footprint

It is estimated that the collected methane generated by each metric ton of food waste in Florida MSW landfills has potential to produce 18.1 kWh of energy. However, the estimated energy required for food waste collection and leachate transportation to a WWTP was 123.2 kWh/Mg and 0.2 kWh/Mg, respectively. Each megagram of waste disposed in a landfill generates approximately 0.23 m^3 of leachate. It requires 3.1 kWh of energy is to treat leachate from each megagram of food waste to the maximum permitted standards before discharging to a receiving water body. The total energy required is therefore approximately 126.5 kWh/Mg while energy expected from landfill gas utilization is around 18.1 kWh/Mg. We estimate a net 108.4 kWh of energy is required to manage each megagram of landfilled food waste in Florida (Table 6), for a total annual energy cost of 1.1 x 10^8 kWh. If every landfill in Florida were collecting and utilizing landfill gas to its full potential ($\varphi_3 = 0.70$), net energy cost after managing each megagram of landfilled food waste is 33.1 kWh, for a total annual energy of 3.24 x 10^7 kWh.

Table 6 Energy consumption and offsets of landfilled food waste

Sector	Energy (kWh/Mg)
Leachate treatment energy cost, E_L	3.1
Food waste Collection energy cost, E_C	123.2
Leachate transport to WWTP energy cost, E_T	0.2
Methane energy production, E_P	18.1
Net energy cost, E	108.4

4.2   Water footprint

It is estimated that up to 58.5 m^3/Mg of high-quality water may be required to dilute treated effluents of leachate attributed to food waste to background concentrations with respect to constituents of concern (Figure 8). In our calculation, mercury (Hg) is the limiting constituent. Concentrations of BOD and TAN in leachate are reduced considerably by treatment. Assuming leachate is treated prior to discharge, up to 3.5 m^3/Mg of water will fully assimilate remaining BOD and TAN concentrations in discharged effluent. However, should untreated leachate be discharged to a water system, up to 63.1 m^3 of water will be required to assimilate BOD and TAN concentrations. The water-energy trade-off for leachate treatment can thus be estimated as 59.6 m^3/Mg of water saved for a net energy requirement of 3.1 kWh/Mg.
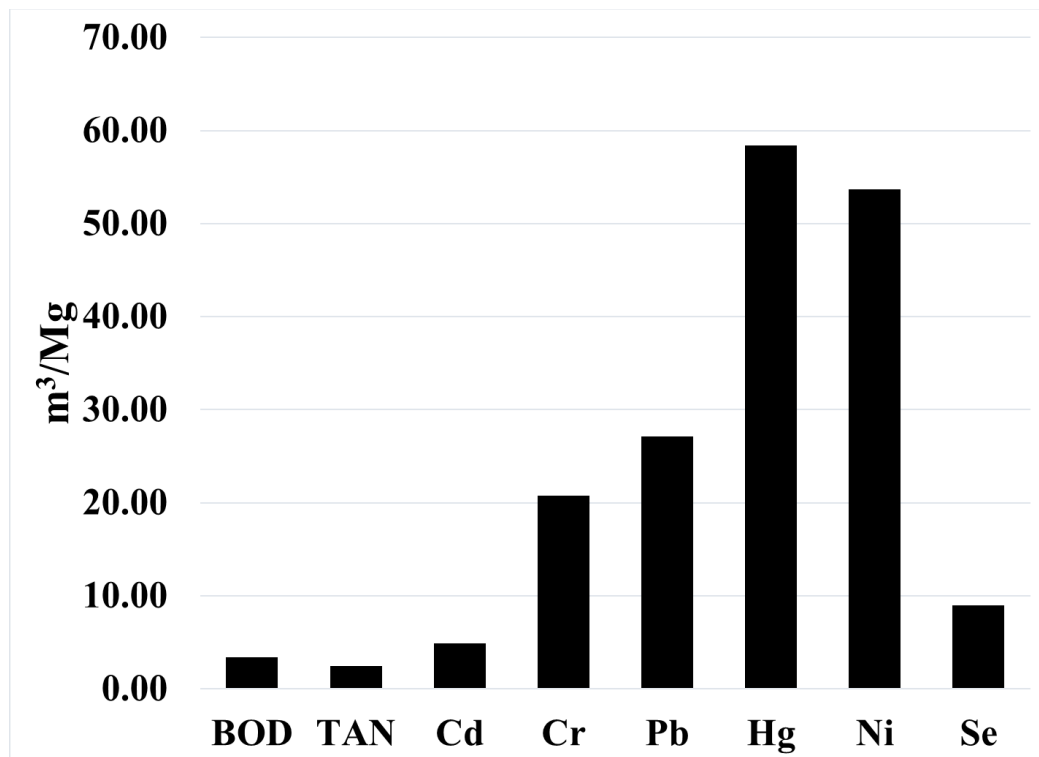
Figure 8: Water footprint of treated food waste leachate
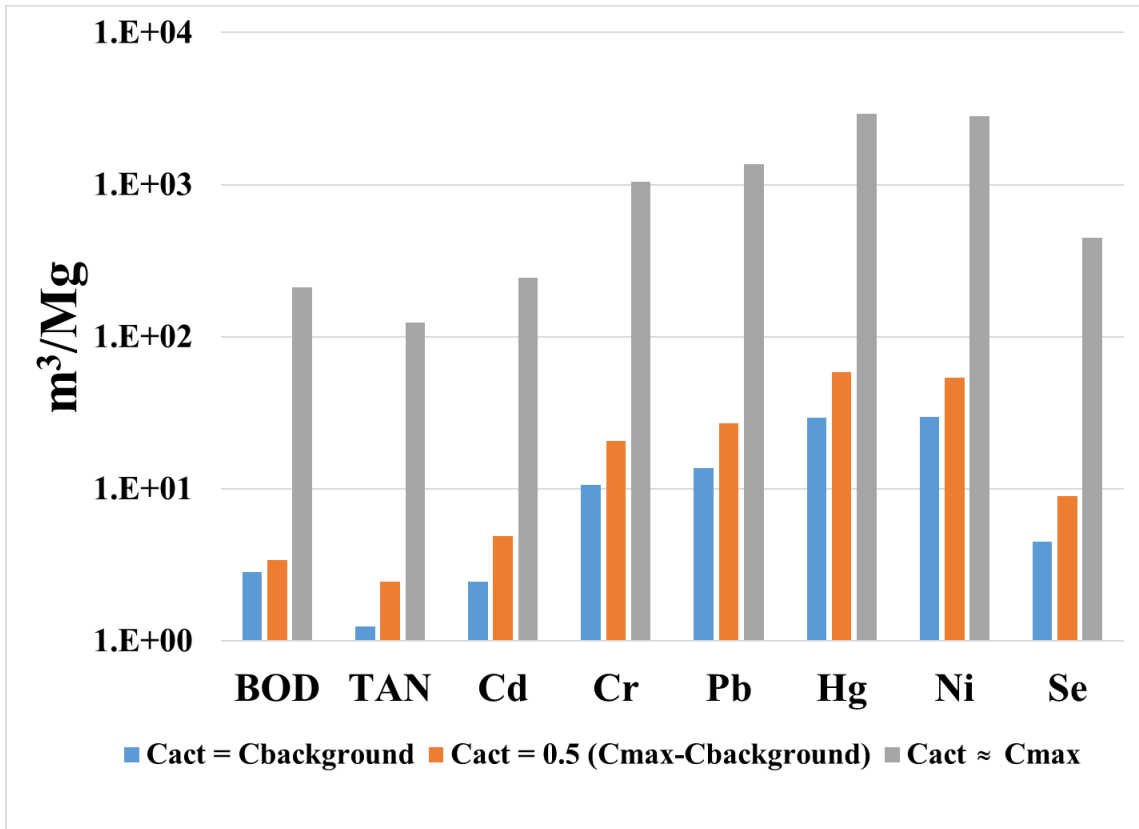
Figure 9 Range of variability in water footprint calculation due to actual concentration in receiving waterbody

Variation of actual concentration creates a significant fluctuation in water footprint calculation (Figure 9). If actual concentration is equal to the background concentration, the water footprint is 29.2 m^3/Mg; mercury is the limiting consituent. If the actual concentration is near to the maximum acceptable concentration, the water cost for waste assimilation becomes 2919 m^3 for each megagram of food waste; indicates the pollution level of a certain water system. The wide range indicates uncertainty in water footprint calculation due to actual concentration.

4.3 Sensitivity analysis

Sensitivity analysis indicates that energy footprints are particularly sensitive to the following parameters: proportionality of actual landfills (φ_3), collection efficiency (φ_1), fuel consumpsion rate for food waste collection (α_{fc}) (Table 7, Figure 10). The net energy footprint across all parameter sets varied from 33.1kWh to 152.0 kWh of energy requirement to manage each megagram of food waste disposed of in a landfill. Whether landfills collect and utilize landfill gas (φ_3, case F) is particularly influential, as energy cost varies from 33.1 kWh/Mg if all landfills harvest gas to 126.5 kWh/Mg of energy loss if none do. The energy model is also sensitive to collection efficiency (η_{ij} ;Case A). The net energy cost ranges from 101.5 kWh/Mg at minimum to 126.5 kWh/Mg at the maximum value. Also, fuel consumpsion rate for food collection (Case:E) points out a fluctuation of 65.5 kWh/Mg of minimum energy cost to 152 kWh/Mg of maximum energy cost. The BOD and TAN concentration in MSW (BOD_{MSW},TAN_{MSW}; Case:G-H) causes a daviation of energy cost from 105.9 kWh to 112.7 kWh to manage each metric ton of food waste. Finally, generation of leachate, which depends on several parameters (Cases: I-L), is somewhat influential, as energy costs range from 105.7 – 111.3 kWh/Mg (Table 7, Figure 10).The model is only marginally sensitive to all other parameters, resulting in maximum ranges of 103.9 -108.9 kWh/Mg.

Also, sensitivity analysis for water footprint indicates that water footprints are particularly sensitive to leachate volume (Table 8, Figure 11). The net water requirement across all parameter sets varied from 8.80 m^3 to 100.1 m^3 to manage each megagram of landfilled food waste. Changing

the background concentration by 30% provides a maximum water footprint of 58.6 m^3 for managing each megagram of food waste after treatment. Permitted concentration and maximum allowable concentrations are regulated by FAC. As discussed, TAN concentrations vary across water bodies and other properties of the water body. The sensitivity of TAN concentration in permitted and allowable concentrations vary from 0.20 m^3 to 3.70 m^3 for managing per megagram of food waste.
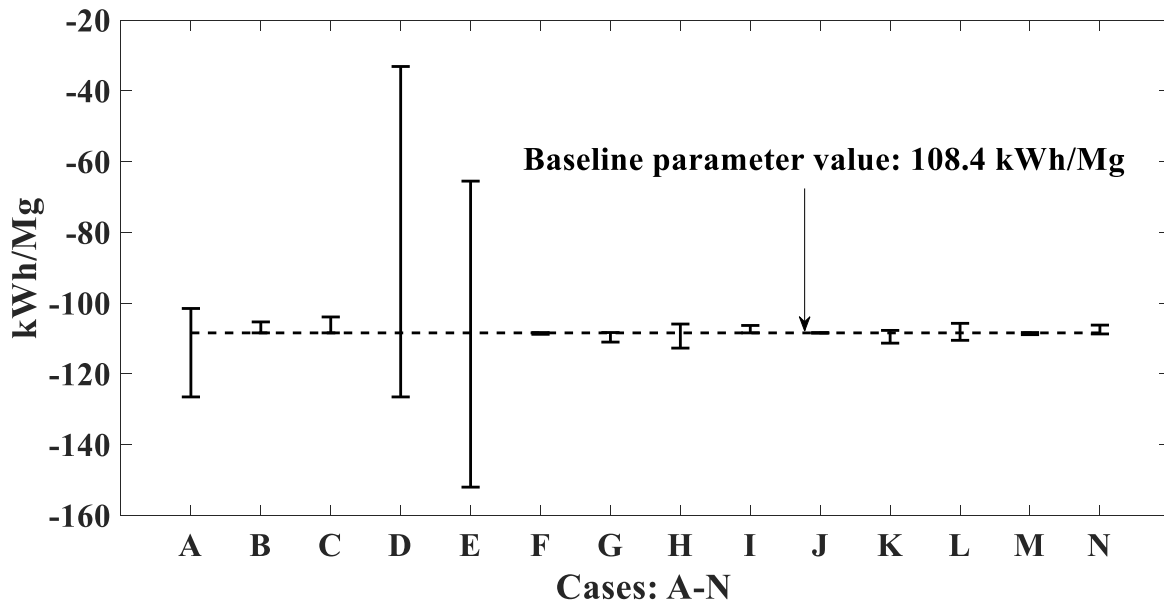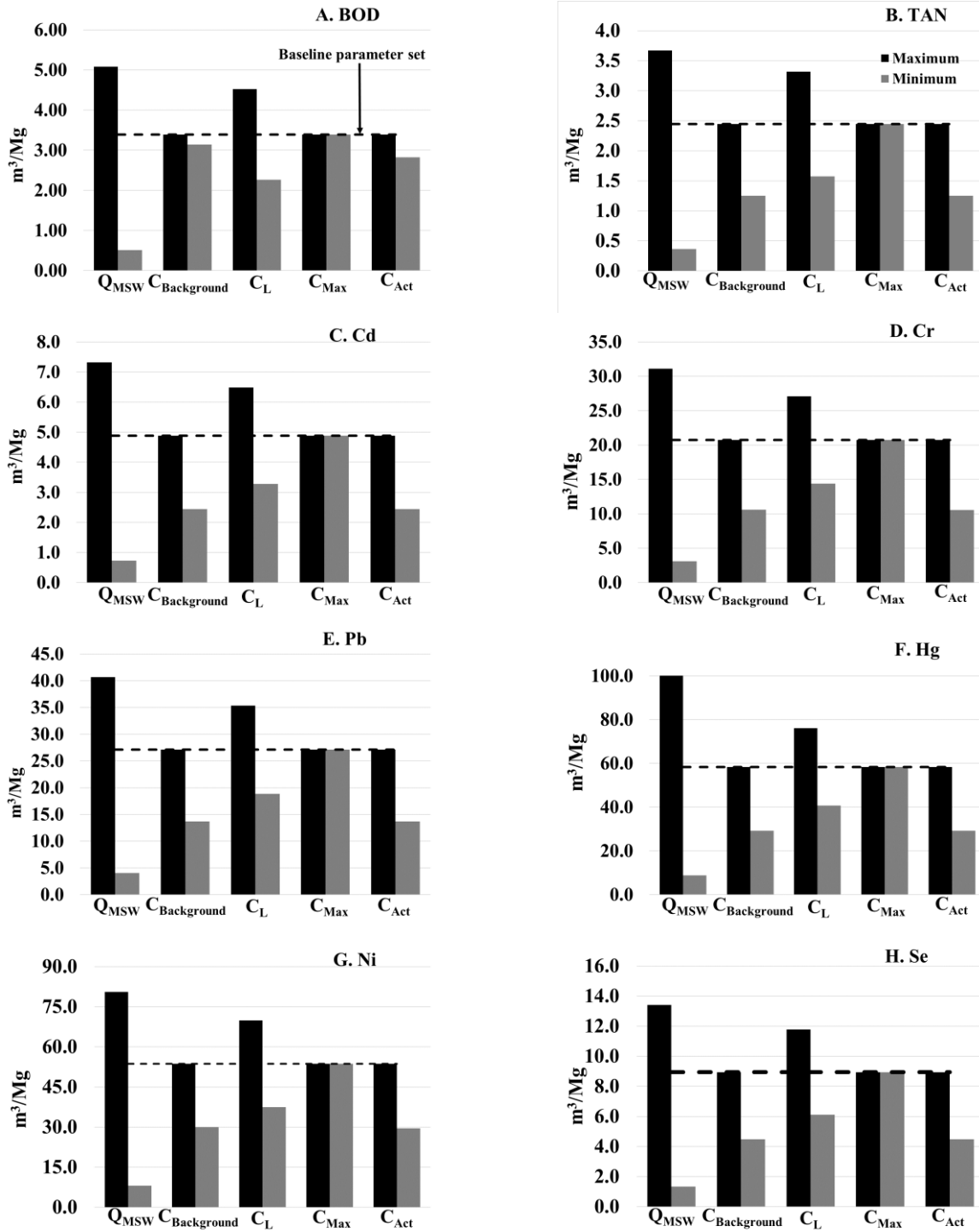

Figure 10: Sensitivity of water footprint parameters

Figure 11: Sensitivity of water footprint parameters

Table 7 Sensitivity of energy footprint parameters

Case	Parameters	Range	Unit	Net energy cost (kWh/Mg)	
				Maximum	**Minimum**
A	Gas collection efficiency (η_{ij})	0-95	Fraction	126.5	101.5
B	Capacity factor (φ_1)	0.83-1	Fraction	108.4	105.3
C	Electrical efficiency (φ_2)	0.35-0.45	Fraction	108.4	103.9
D	Proportion of landfills collecting LFG (φ_3)	0-0.70	Fraction	126.8	33.1
E	Fuel consumption rate, food waste collection (α_{fc})	7.2-15	L/Mg	152.0	65.5
F	Fuel consumption rate, leachate transport (α_{fT})	0.79-1.02	L/Mg	108.8	108.3
G	BOD in MSW leachate (BOD_{MSW})	68-3730	mg/L	111.0	108.3
H	TAN in MSW leachate (TAN_{MSW})	98-2300	mg/L	112.7	105.9
I	Fraction of precipitation collected as leachate (F_t)	6.5-20	%	108.4	106.3
J	Precipitation (P) in Florida	1085-1350	mm	109.0	108.3
K	Density of MSW (ρ)	413-1003	kg/m^3	111.3	107.7
L	Waste Depth (D)	10-100	m	110.5	105.7
M	O_2 requirement for BOD treatment (k_1)	1.4-2.0	kg/kg	108.9	108.3
N	O_2 requirement for TAN treatment (k_2)	4.0-5.0	kg/kg	108.7	106.2

Table 8 Sensitivity of water footprint parameters

Parameter/constituent	Range	Unit	Water footprint (m^3/Mg)	
			Minimum	Maximum
Leachate generation rate, $Q_{MSW,T}$	0.035-0.404	m^3/Mg		
BOD	--	--	0.5	5.8
TAN	--	--	0.4	4.2
Cadmium	--	--	0.7	8.4
Chromium	--	--	3.1	35.6
Lead	--	--	4.1	46.5
Mercury	--	--	8.8	100.1
Nickel	--	--	8.1	92.0
Selenium	--	--	1.3	15.3
Background concentration, $C_{background,i}$				
BOD	0.35-0.65	mg/L	3.1	3.4
TAN	10.5 – 19.5	µg/L	1.3	2.4
Cadmium	0.0007 – 0.0013	µg/L	2.4	4.9
Chromium	0.07-0.13	µg/L	10.6	20.7
Lead	0.028-0.052	µg/L	13.7	27.1
Mercury	0	mg/L	29.2	58.6
Nickel	0.28-0.52	µg/L	30.0	53.7
Selenium	0	mg/L	4.5	8.9
Permitted concentration, $C_{L,i}$				
BOD	14-26	mg/L	2.3	4.5
TAN	8.75-10	mg/L	1.6	3.3
Cadmium	0.07-0.13	mg/L	3.3	6.5
Chromium	0.35-0.65	mg/L	14.4	27.1
Lead	0.35-0.65	mg/L	18.8	35.4
Mercury	1.05-1.95	µg/L	40.7	76.0
Nickel	0.7-1.3	mg/L	37.4	69.9
Selenium	0.07-0.13	mg/L	6.1	11.8

4.4   Uncertainty analysis

The simulation indicates that the water and energy footprint are normally distributed (Figure 12). Different statistical parameters are shown in the Table 9. The analysis shows that the estimated net energy cost using the baseline parameter set is 28% more than the average value and the estimated water footprint using the baseline parameter set is 2% less than the average value calculated from the Monte Carlo simulation. This approach was applied to see the effect of uncertainty on the system. Further research should include more advanced approach to integrate the uncertainty within the FEW nexus such as quantifying and characterizing the data set and its distribution, advanced algorithm in addressing uncertainty etc.
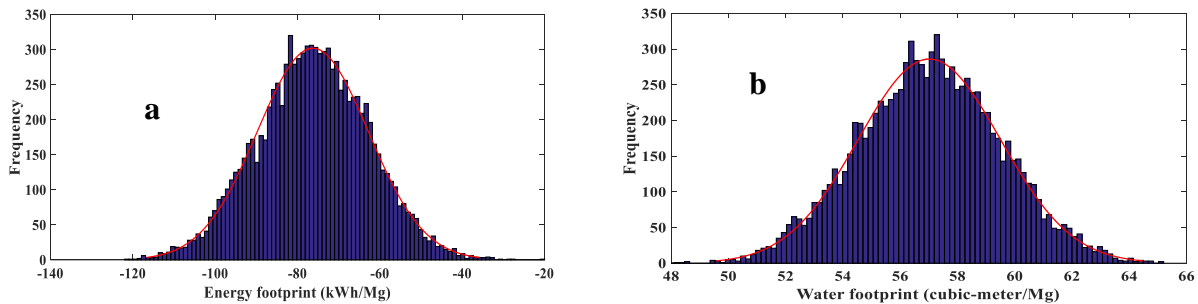
Figure 12 Energy (a) and water (b) footprint (Monte-Carlo Simulation)

Table 9 Statistics of simulated results from Monte-Carlo simulation

	Net energy cost (kWh/Mg)	Net water cost (m³/Mg)
Maximum	121	67
Minimum	20	47
Average	76	57

The uncertainty analysis, introduced here is an attempt to address within the FEW nexus. There are some limitations in the assessment. In this analysis, it is assumed that all the sensitive parameters are uniformly distributed. But the distribution of these parameters is unknown. It is

required to define the distribution using a large dataset. Also, more advanced techniques need to

defined for this system.

CHAPTER FIVE: DISCUSSION

The estimated annual energy cost to Florida's energy sector is 1.1×10^8 kWh. Per capita energy cost is nearly 5.0 kWh/year. This represents 0.05% of Florida's total energy generation (USEIA, 2016), or around 2% of Florida's 2015 renewable energy production (USEIA, 2016). In terms of energy consumption, the estimated energy cost of landfilled food waste is 0.03% of Florida's total residential energy consumption (USEIA, 2016).

The annual volume of water required for assimilating food waste leachate in Florida is up to 5.8×10^7 m^3. At 2.9 m^3 of water per capita, this is 0.67% of the per capita water consumption in Florida (440m^3 in 2012, USGS, 2016) and nearly 0.65% of the total freshwater withdrawn in Florida in 2012.

5.1 Energy and water footprint of wasted food and food waste management

The estimates of the post-disposal energy and water impacts of food waste allow for the first comprehensive estimate of food waste impact within the FEW nexus, considering resources needed both in producing wasted food and managing food waste after disposal. A total energy impact is estimated as 1.4×10^{10} kWh to produce if wasted food is managed by landfill disposal. The energy footprint of landfill disposal varies nearly 0.76 to 1.16% of the energy needed to produce wasted food.

Water footprint is also calculated if food waste is managed by landfill disposal. The water impact is nearly 1.5×10^{10} m^3 to produce. The estimated water footprint of managing wasted food varies nearly 0.40% to 1.75% of the water needed to produce wasted food.

5.2 Sensitivity analysis

Sensitivity analysis indicated that the landfill contribution to energy production and contaminant loading affects the water and energy footprint. Only 13% of Florida landfills producing landfill gas and contributes to the energy production. If full utilization (70%) of landfill gas is considered, it is found that nearly 72% decrease in energy needed to manage landfilled food waste. Even in the full utilization scenario, the energy potential of landfilled food waste does not fully offset consumption.

Contaminant loadings are determined by the combination of leachate generation rate and constituent (e.g. BOD and TAN) concentrations attributable to food waste. Among the parameters controlling leachate generation rate, density and annual precipitation vary the leachate generation rate significantly. It was not tested about the influence of other parameters (temperature, soils etc.) which may affect the leachate generation rate (Maimoun, 2015), given the relative uniformity of these factors within the system boundaries. Concentrations of constituents in MSW leachate that are attributable to food waste cannot be observed directly. To estimate the contribution of food waste to MSW leachate concentrations, this research includes scaled concentrations of BOD and TAN observed in MSW leachate per carbon and nitrogen proportionality.

5.3 Water energy tradeoff

Using the FEW nexus framework, tradeoffs in water and energy with regard to leachate treatment can be estimated. For example, up to 3.5 m^3 of high quality water is required to assimilate BOD and TAN from each metric ton of food waste after leachate is treated at a WWTP. If leachate is not treated, up to 63.1 m^3 of water per metric ton of food waste is required for assimilation of carbon and nutrients in food waste. Therefore, leachate treatment can ideally save up to 59.6 m^3

of water at the cost of 3.1 kWh of energy in each metric ton of food waste. The water footprint of this quantity of energy is negligible (0.0031 m^3) assuming WWTPs are operated by electricity produced by natural gas (USEIA, 2016) and the water footprint of gas produced energy is 1000 m^3/kWh (DHI, 2008). The water footprint of energy required for leachate treatment is smaller than the water footprint of direct discharge to the environment. Beyond validating the rationality of wastewater treatment, the energy and water tradeoff presented can potentially inform valuation of waste assimilation as an ecosystem service.

5.4   Water for pollution assimilation

The grey water footprint returns the volume of water needed to fully assimilate a pollutant load, and therefore does not represent a fully consumptive use of water. However, direct comparison of grey water footprints with consumptive uses (blue and green water footprints) is common. For example, Shao and Chen, 2013 compared the grey and blue water footprints in China, finding that the water required for waste assimilation exceeded consumptive uses. Liu et al., 2012 introduced the grey water footprint for anthropogenic emissions to the waterbody and describes the impact on how grey water footprint can impact the freshwater withdrawal. Morera et al., 2016 has identified the grey water contribution as the significant water use in a wastewater treatment plant as compared to green and blue water. Impact of the grey water footprint is perhaps best conceptualized through an ecosystem services framework. For instance, it is estimated that 59.6 m^3 of grey water can be offset by 3.1 kWh of electricity for treating nutrients associated with each metric ton of landfilled food waste. Assuming the cost of commercial energy in Florida is $0.08/kWh. (USEIA, 2016) and applying our derived energy-water tradeoff, we estimate an assimilation service value of around $0.004/$m^3$ per cubic meter of assimilation water. The annual

ecosystem value of assimilating nutrients from landfilled food waste in Florida is therefore around $0.22 million. This rough valuation via an energy-water tradeoff likely underestimates the value of assimilation water, as the valuation may not take into account true energy costs, nor the capital cost of treatment infrastructure. The value of assimilation water estimated by a water-energy tradeoff is low as compared to values estimated globally. For example, Costanza et al., 1997 has estimated the value of wastewater assimilation as $0.59/m^3$; assuming 1m of water height and considering the inflation rate from 1997 to 2017 (US Inflation Calculator, 2017). Using this valuation, it is estimated that the annual ecosystem value of assimilating nutrients from landfilled food waste in Florida as $35 Million.

5.5 Grey water footprint

The formulation grey water footprint calculation is hard to comceptualize. Different researchers has identified grey water footprint in different approaches. Hoekstra et al., 2009 has used background concentration to find out the assimilation capacity. Again, Hoekstra et al., 2011 has introduced actural concentration to calculate the grey water footprint. Morera et al. 2016, has introduced mass-balance concept to find the grey water for waste water treatment plant but limited to use the actual concentration in the receiving water. These three different approaches are difficult and introduce ambiguity in grey water footprint calculation. Moreover, the actual concentration in grey water footprint calculation is distrinct in pattern from one waterbody to another and varies temporally. The actual concentration needs to be addressed properly while calculating the grey waterfootprint for a particular region.

5.6   FEW implications for food waste management

Herein it is estimated that the post-disposal FEW impact deriving from the portion of Florida's food waste that is managed by landfill disposal, which is around 54% of Florida's food waste (FDEP, 2014). The remainder is managed via alternative waste management technologies which include composting, anaerobic digestion, and thermal conversion. The FEW impacts of these alternative technologies have not been estimated, and may differ from those of landfilling. It is possible that further analyses of FEW impact could inform societal decision making and investments regarding food waste management technologies. For instance, should subsequent analyses characterize the FEW impact of waste management alternatives, comparison of technologies could suggest best management alternatives for food waste based on minimizing FEW impacts.

CHAPTER SIX: FUTURE STUDIES

In this analysis, energy and water tradeoffs were calculated for landfilling, a traditional approach for food waste management. A parallel approach to addressing the water and energy costs of food waste could involve changing the management paradigm for organics in the solid waste stream. For instance, alternative pathways for food waste nutrients and carbon could include diversion from landfills to composting or energy generation through anaerobic digestion. However, best management practices towards food waste management require a complete understanding of food waste within the FEW nexus. Specifically, information comparing the water costs of various management alternatives which does not exist. Future studies should address a framework for other food waste management techniques and then quantify the water and energy footprint.

The water footprint calculated in this research did not include the water which is required for energy production used in transporting leachate and waste and treating leachate . Only pollutant assimilation is considered in the water footprint calculation but water is required to produce any form of energy (Lundie and Peters, 2005; Finnveden et al., 2005; Berglund and Borjesson., 2006). Further study should extend system boundaries to include the water footprint of energy used in food waste management.

There is undeniable uncertainty associated with the method of estimation for BOD and TAN concentration in food waste stream. Given the importance of leachate concentrations to both energy and water footprints, it is recommended further research to more precisely characterize the likely contribution of contaminants to MSW leachate.

In this research, ecosystem service is addressed through service cost of water and energy and compared with global estimate of waste water assimilation. Future research should address the detailed ecosystem service framework of wastewater assimilation within the FEW nexus.

CHAPTER SEVEN: CONCLUSIONS

To address deficiencies in understanding food waste within the FEW nexus, this research made the first attempt at quantifying food waste impacts related to post-disposal management and estimating the energy and water needed to manage Florida's landfilled food waste. Food waste management by landfilling in Florida consumes around 5.8×10^7 m^3 of water (0.7% of annual fresh water withdrawal in Florida) in 2014 and 1.1×10^8 kWh (0.05% of annual Florida electricity production) in 2014. This estimation of water and energy fluxes was found to be sensitive to landfill gas utilization rates, volumes of MSW leachate generated, and the concentration of contaminants in leachate that is derive from food waste. These parameters should be carefully characterized in future FEW studies.

The estimates of the post-disposal energy and water impacts of food waste allows for the first comprehensive estimate of food waste impact within the FEW nexus, considering resources needed both in producing wasted food and managing food waste after disposal. It was found that resources needed to manage landfilled food waste varies from up to 1.16% of the energy and up to 1.75% of water needed to produce the wasted food. Society can perhaps reduce the impact of wasted food by reducing food waste or by choosing more FEW-efficient management technologies. Further studies and comparisons of the FEW impacts related to alternative food waste management technologies (e.g. composting, thermal conversion, anaerobic digestion) have the potential to guide investments and decisions related to more sustainable food waste management.

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