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Life-cycle Greenhouse Gas Emissions and Water Footprint of Residential Waste Collection and Management Systems

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LIFE-CYCLE GREENHOUSE GAS EMISSIONS AND WATER FOOTPRINT OF RESIDENTIAL WASTE COLLECTION AND MANAGEMENT SYSTEMS

by

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ABSTRACT

Three troublesome issues concerning residential curbside collection (RCC) and municipal solid waste (MSW) management systems in the United States motivated this research. First, reliance upon inefficient collection and scheduling procedures negatively affect RCC efficiency, greenhouse gas (GHG) emissions, and cost. Second, the neglected impact of MSW management practices on water resources. Third, the implications of alternative fuels on the environmental and financial performance of waste collection where fuel plays a significant rule.

The **goal** of this study was to select the best RCC program, MSW management practice, and collection fuel. For this study, field data were collected for RCC programs across the State of Florida. The garbage and recyclables generation rates were compared based on garbage collection frequency and use of dual-stream (DS) or single-stream (SS) recyclables collection system. The assessment of the collection programs was evaluated based on GHG emissions, while for the first time, the water footprint (WFP) was calculated for the most commonly used MSW management practices namely landfilling, combustion, and recycling. In comparing alternative collection fuels, two multi-criteria decision analysis (MCDA) tools, TOPSIS and SAW, were used to rank fuel alternatives for the waste collection industry with respect to a multi-level environmental and financial decision matrix.

The results showed that SS collection systems exhibited more than a two-fold increase in recyclables generation rates, and a ~2.2-fold greater recycling efficiency compared to DS. The GHG emissions associated with the studied collection programs were estimated to be between 36 and 51 kg CO_{2eq} per metric ton of total household waste (garbage and recyclables), depending on the garbage collection frequency, recyclables collection system (DS or SS) and recyclables compaction. When recyclables offsets were considered, the GHG emissions associated with

programs using SS were estimated between -760 and -560, compared to between -270 and -210 kg CO_2eq per metric ton of total waste for DS programs. In comparing the WFP of MSW management practices, the results showed that the WFP of waste landfilling can be reduced through implementing bioreactor landfilling. The WFP of electricity generated from waste combustion was less than the electricity from landfill gas. Overall, the WFP of electricity from MSW management practices was drastically less than some renewable energy sources. In comparing the WFP offsets of recyclables, the recycling of renewable commodities, e.g. paper, contributed to the highest WFP offsets compared to other commodities, mainly due to its raw material acquisition high WFPs. This suggests that recycling of renewable goods is the best management practice to reduce the WFP of MSW management. Finally, the MCDA of alternative fuel technologies revealed that diesel is still the best option, followed by hydraulic-hybrid waste collection vehicles (WCVs), then landfill gas (LFG) sourced natural gas, fossil natural gas and biodiesel. The elimination of the fueling station criterion from the financial criteria ranked LFG-sourced natural gas as the best option; suggesting that LFG sourced natural gas is the best alternative to fuel WCV when accessible.

In conclusion, field data suggest that RCC system design can significantly impact recyclables generation rate and efficiency, and consequently determine environmental and economic impact of collection systems. The WFP concept was suggested as a method to systematically assess the impact of MSW management practices on water resources. A careful consideration of the WFP of MSW management practices and energy recovered from MSW management facilities is essential for the sustainable appropriation of water resources and development.

To my parents Wafa and Awad

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1: INTRODUCTION

Municipal solid waste (MSW) is generated by daily activities at homes, hospitals, schools, businesses, and industries (U.S. EPA, 2014a). MSW has two sources, residential (in the U.S., 55-65% of total MSW) and commercial (35-45% of total MSW) (Smith, 2012). Residential waste is collected from single or multi-families dwellings. A single family residence is an individual structure with its own lot and is usually serviced by residential curbside collection (RCC), whereas multi-family residences are connected structures and are usually provided with dumpsters. The main focus of this research is RCC, which includes over 8,660 programs throughout the U.S. (Smith, 2012), each usually providing garbage, recyclables, yard waste, and in some cases, food waste, collection lines (Figure 1.1). Such service necessitates a minimum of three weekly collection lines. These collection schedules persist over the entire year for public convenience, although waste generation rates and collection needs vary seasonally, e.g., holiday and low-growth vegetation seasons (Maimoun et al., 2013).

The design of RCC programs varies significantly among U.S. communities; major differences include number of collection lines provided; collection frequency; type of recycling collection system (single-stream or dual-collection); the number, type and volume of garbage and recycling containers; and fuel used. As municipalities aim to reduce the environmental and financial impacts of recyclables collection while increasing customer satisfaction, optimized design of the RCC system will be a first step toward achieving sustainable waste management. Accordingly, this research was designed to explore trade-offs between environmental and economic factors to ensure sustainable operation of RCC systems which could lead to lower cost, more convenient RCC programs at minimal environmental impact, e.g., more recycling, less landfilling or combustion, and avoided use of new resources.

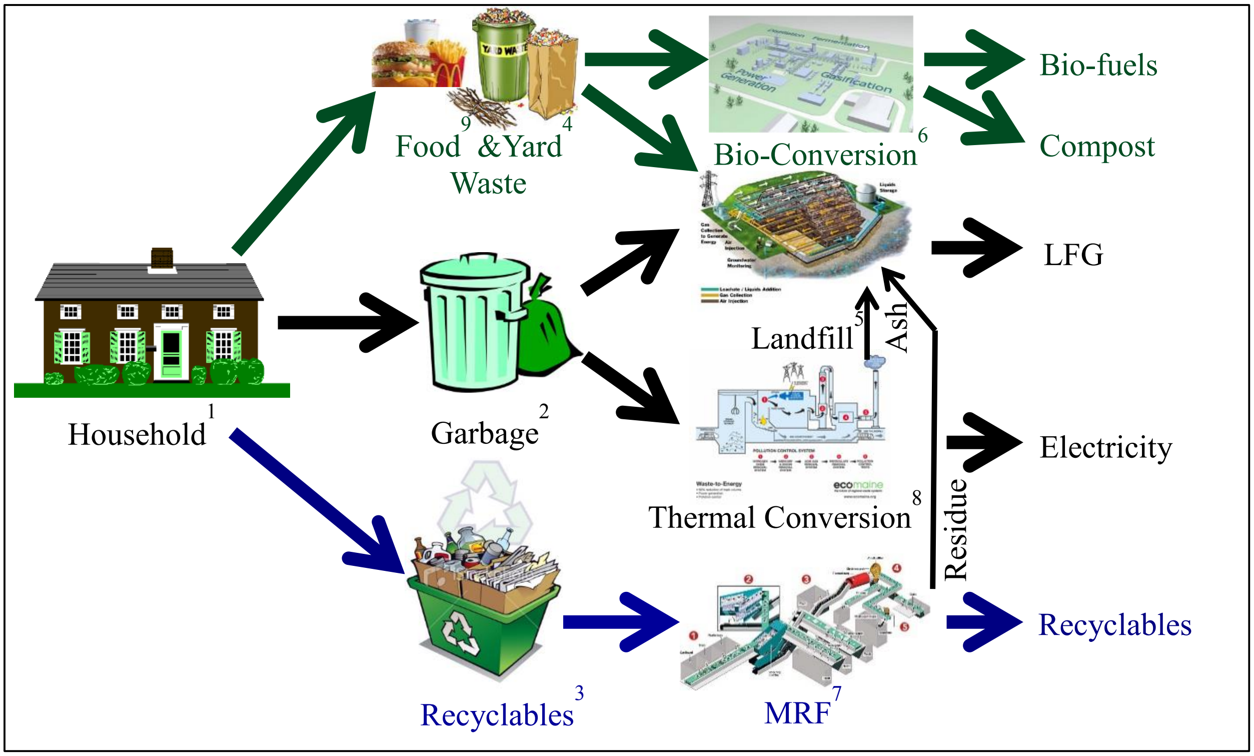

Figure 1.1: Residential Waste Collection and Management systems (Graphic Sources: 1) North Dakota Department of Health; 2) krmsradio; 3) Recycle Bin, Logo & Recyclables | Stock Illustration | iStock; 4) City of Waltham, Massachusetts; 5) Waste Management, Inc.; 6) INESO Bio; 7) Reid Brown; 8)Ecomaine; 9) McDonalds).

RCC programs are a part of the overall municipal solid waste (MSW) management system that manages, treats and disposes municipal waste. In general, MSW management practice can be classified into four major categories: (1) landfilling, (2) recycling, (3) bio-conversion of organic wastes to products, and (4) thermal conversion (Figure 1.1). In 2012, about 135 million tons of the U.S. MSW (53.8% of total generated) were discarded in landfills, and about 34.5 percent was recycled (U.S. EPA, 2014a); ranking waste landfilling and recycling as leading MSW management practices in the U.S.. The effectiveness of MSW management practices has been evaluated in the published literature and assessment models based on their greenhouse gas (GHG) emissions; economic costs; and airborne, soil, and waterborne emissions (Weitz et al., 1999; Weitz et al., 2002; Consonni et al., 2005; Winkler and Bilitewski, 2007; Buttol et al., 2007; Cherubini et al., 2009). However, the direct and indirect impacts of MSW management practices on water resources have been neglected or not fully considered. During the last decade, the water footprint (WFP) methodology (Hoekstra, 2003; Chapagain and Hoekstra, 2004; Hoekstra et al., 2009) has been developed and used to capture and quantify both direct and indirect effects of processes, products, entities, industries, energy sources, and countries on water resources (Chapagaina et al., 2006; Chapagain and Hoekstra, 2007; Dominguez-Faus et al., 2009; Gerbens-Leenesa et al., 2009a; Gerbens-Leenesa et al., 2009b; Hadian and Madani, 2013). Some major industries, entities, and corporations are using the WFP concept as a tool for sustainable appropriation of freshwater resources (Pahlow and Mekonnen, 2012). The WFP provides a reliable criterion for evaluating water use efficiency (Hadian and Madani, 2013) by measuring both the direct and indirect use of fresh water over the entire process life-cycle (Hoekstra et al., 2009).

Waste collection accounts for 40-60% of the total MSW management budget (Bueno, 2011). A major cost element of any MSW collection and management system is the transportation fuel; historically, the operation of RCC has been reliant on diesel fuel waste collection vehicles. In fact, prior to 2009, diesel-fueled waste collection vehicles were the backbone of the waste collection industry with less than one percent using alternative fuels (Rogoff et al., 2009). After 2010, relatively low prices of natural gas compared to high unstable diesel prices have increased industry interest in natural gas waste collection vehicles (Maimoun et al., 2013). By 2012, waste collection vehicles and transfer vehicles accounted for 11 percent of the total U.S. natural gas vehicles (NGVAMERICA, 2012). Diesel fuel purchase was estimated to consume 7.5% of the industry revenues in 2012 (Smith, 2012).

Undoubtedly, the driving factor for the recent waste industry switch to natural gas is fuel cost. However, a comprehensive decision matrix that considers other factors as well as changing policies, future fuel prices and uncertain fuel performance data, has not been considered. In the last three decades, the selection scheme for alternative fuels and energies has changed from a single-criterion cost-based assessment, to a multi-criteria analysis that considers environmental, social, operational, and political factors (Pohekar and Ramachandran, 2004; Cavallaro , 2005; Wang et al., 2009; Linkov and Moberg , 2011).

1.1.Objectives and research questions

The **goal** of this study was to select the best RCC program, management practice, and collection fuel. The specific three objectives of this project were to:

(1) **Evaluate the recycling efficiency, GHG emissions, and collection cost of RCC programs as a function of recycling participation rate, collection frequency, and collection system design (dual or single stream).** This objective aims to address the following research questions:

- What is the effect of the RCC system design on waste generation rates and recycling efficiency, which in turn affects waste management cost and environmental impacts?
- How will increasing recycling participation rate affect environmental and financial benefits of the recyclable curbside collection?

(2) **Evaluate the WFP of commonly used MSW management practices, namely: landfilling, waste combustion, and recycling.** This objective aims to address the following research questions:

- What is the impact of MSW management practices on water resources, and how it can be minimized?
- What is the WFP of energy products and recycled commodities diverted from landfills?

(3) Perform a multi-criteria decision analysis (MCDA) of alternative fuel waste collection vehicles. This objective aims to address the following research questions:

- What is the best alternative fuel for waste collection vehicles?
- What will be the effect of future policies and fuel economics on alternative fuel selection?
- What are the advantages, if any, provided by green fuels, i.e., biodiesel and LFG compared to conventional fossil fuels?

1.2. Dissertation outline

A dissertation outline is as follows: Chapter 2 reviews the current literature on RCC programs and identifies the research gaps that will be addressed. In Chapter 3, field data were collected for RCC programs across Central Florida. The effect of the RCC system design on waste generation rates and recycling efficiency was assessed. The environmental impacts of waste collected under different RCC system designs was evaluated based on life-cycle GHG emissions, while the economic performance was evaluated based on collection cost per ton of waste and, in some cases, financial benefits from selling any by-products. This chapter was written as a manuscript entitled "*An Environmental-Economic Assessment of Residential Curbside Collection Programs in Florida"* and submitted to the *Resources Conservation and Recycling Journal*.

In Chapter 4, a comprehensive WFP calculation methodology for evaluating the effects of MSW management practices on water resources is described. The local and global WFPs of the three most commonly used MSW management practices (landfilling, combustion with energy recovery, and recycling) were then determined and compared. The calculated WFPs of MSW management practices were compared to their other environmental burdens (e.g. GHG emissions). This chapter was written as a manuscript entitled *"The Water Footprint of Common Municipal*

Solid Waste Management Practices” and will be submitted to the *Environmental Science and Technology Journal*.

In Chapter 5, two multi-criteria decision making (MCDM) methods were used to rank fuel alternatives for waste collection vehicles with respect to a multi-level environmental and financial decision matrix. A sensitivity analysis was conducted to evaluate the robustness of the results to the selection criteria and future energy pricing scenarios. This chapter was written as a third manuscript entitled *“Multi-level Multi-criteria Analysis of Alternative-fuels for Waste Collection Vehicles in the United States”* and will be submitted to *Science of the Total Environment Journal*. Chapter 6 presents conclusions and recommendations drawn from this research. A flowchart connecting the main goal of this dissertation with objectives and outcomes is shown in Figure 1.2.

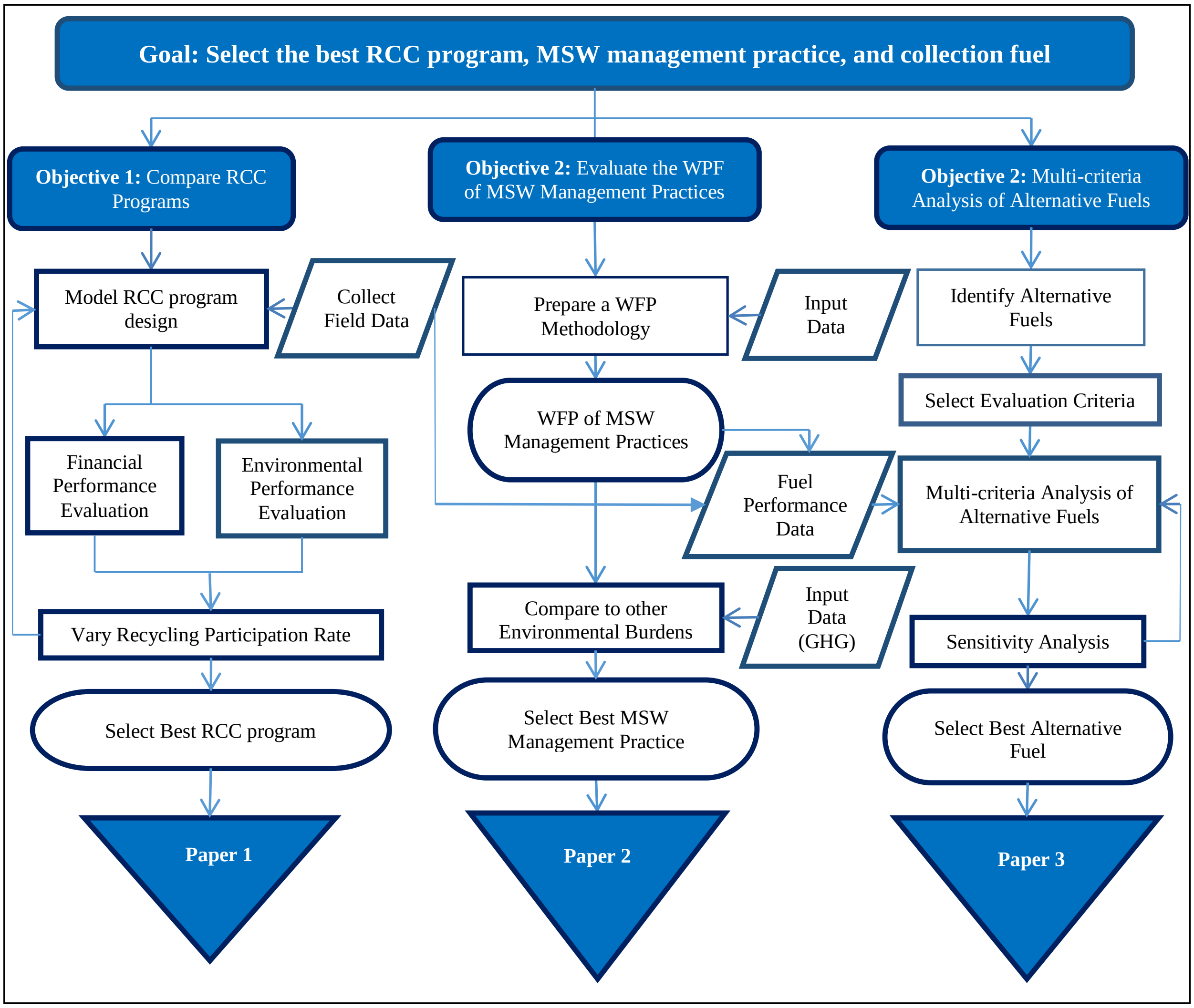

Figure 1.2: Dissertation Flowchart.

CHAPTER 2: LITERATURE REVIEW

2.1. Residential curbside collection lines

The majority of the US RCC programs includes garbage, recyclables, yard waste, and in some cases, food waste, collection lines. The collection frequency of garbage varies based on the climate, competition, topography and the price of service (Sahoo et al., 2005; Kim et al., 2006). In the past, the northern part of the U.S. was served once weekly, whereas the southern part of the U.S. was served twice weekly to reduce odors (Kim et al., 2006). However, RCC programs are faced with rising collection costs and diversion of waste to recyclable and yard waste lines, providing impetus to switch to once per week or every other week (bi-weekly) waste collection. On the other hand, the main disadvantage of reducing waste collection frequency to weekly or bi-weekly is the health concerns associated with leaving food waste in bins for up to two weeks (McLeod and Cherrett, 2008). According to the U.S. EPA (2014a), food waste was 14.5 percent of the MSW stream generated in the U.S. in 2012, and it was the largest component of MSW discards after recycling and recovery (U.S. EPA, 2014a). The curbside collection of food waste as a separate stream is less common in the U.S. in comparison with Europe (SWANA, 2008). In the U.S., food wastes are either thrown out in the garbage stream (SWANA, 2008); or discarded using food waste disposal units (Iacovidou et al., 2012a); or in few cases food waste was collected mixed with yard waste (SWANA, 2008). UK researchers have recommended that food waste should be collected separately on weekly basis to divert waste from landfills (WRAP, 2007). In 2008, there were less than 100 communities in the U.S. served by curbside collection of food waste (SWANA, 2008). Most of these collection programs were established in the last decade, and 56 of these communities were in only four states (SWANA, 2008). In these communities, food waste was either collected by itself or mixed with yard wastes, on a weekly or bi-weekly collection basis (SWANA, 2008).

In 2010, it was estimated that over 97% of food waste in the US was disposed in landfills (Levis et al., 2010), although approximately 50% of the US households have food waste disposal units (CECED, 2003; U.S. EPA, 2008a; Iacovidou et al., 2012b), which is by far the highest worldwide (Iacovidou et al., 2012a).

After years of public education, most households in the US understand the importance of recycling in conserving resources and reducing the waste stream. However, customer convenience plays a vital role in the amount of the recovered material. Everett and Peirce (1993) studied the effect of collection frequency, collection day, and providing containers on the material recovery rate (MRR) by voluntary and mandatory curbside recycling programs. The study concluded that providing containers slightly improved curbside MRR for voluntary collection program, but not mandatory programs. On the other hand, increasing recyclables collection frequency had a slightly positive effect on the MRR, while collection day had only a slight effect on the MRR (Everett and Peirce, 1993). In 2009, there were 9,066 curbside recycling programs nationwide, up from 8,875 in 2002 (U.S. EPA, 2011a; BioCycle, 2006); 71% of the US population was served by a RCC in 2010 (U.S. EPA, 2011a). According to the American Beverage Association (2009), approximately 228 million Americans, or 74 percent of the total population, had access to curbside recycling, while 83% of the total population had access to drop-off recycling facilities.

According to the U.S. EPA (2011a), the implementation of curbside collection of recyclables is expected to increase recycling; diverting reusable materials from the waste stream. Lave et al. (1999) argued that for most MSW recycling categories the costs of collection and processing were expected to exceed the avoided disposal fee and revenues from sales of recyclables. In their calculation, they assumed an extra collection cost for the recycling service line with no changes to the waste collection cost. However, recycling diverts substantial quantities of material from the waste stream; therefore, savings in waste collection are expected.

In the U.S., RCC programs can be classified according to the number of collection streams; dual-stream or single-stream. Dual collection requires residents to separate cardboards, papers, and magazines from the rest of recyclable materials using 60-liter (16-gallon) bins, while single stream collection allows residents to mix all recyclable material together using 60-liter (16-gallon) to 240-liter (64-gallon) containers. The number of containers provided for residents varies based on the collection system used and the hauling contract. In general, residents are not willing to use more than two bins (Personal Communication with Alan Morrison, 2012).

During the last decade, many communities in the U.S. have switched from dual collection to single-stream recycling due to the ease of operation (Fitzgerald et al., 2012). An average of 14 new single-stream material recovery facilities (MRFs) have been added every year since 1995 (Fitzgerald et al., 2012; Berenyi, 2008). Fitzgerald et al. (2012) examined the quantities of recycled material at three MRFs, and they concluded that switching from dual collection to single stream generated 50% more recyclables. In their analysis, data obtained from two of the facilities were collected two years apart. One of the compared MRFs did not accept paper recycling before switching to single-stream recycling. Therefore, the tonnage increase as a result of switching to single-stream ranged from 23-85% among the three MRFs (Fitzgerald et al., 2012). The analysis of recovered material at MRFs did not account for the effect of the collection service frequency, type or number of recycling containers provided on the recycling efficiency.

In the UK, Williams and Cole (2013) examined the effect of alternate weekly collection (AWC) of residential waste and recyclables on recycling quantities. The study also evaluated the impact of the number of streams on the recycling efficiency. The authors concluded that the AWC schemes positively impacted the recycling rate without any adverse impact on public participation. In order to compare dual stream and single stream, two experiments were conducted. Both experiments involved a reduction in recyclables collection frequency from weekly to biweekly, while changing the recyclables collection streams to either dual-stream or single-stream. In comparing the experiments, dual stream performed better than the single stream, collecting an average of 5.94 kg/household/week compared to an average of 5.63 kg/ household /week. However, the single stream showed a greater increase in the weight of material collected compared to the same period before the experiment (0.53 kg/household/week increase for single-stream compared to 0.48 kg/household/week for dual-stream). The authors concluded that the single stream collection is better than dual collection based on evaluation criteria.

Finally, the majority of RCC programs includes a third service line for yard waste collection. According to the U.S. EPA (2014a), yard waste accounted for 13.5 percent of the MSW generated in the US in 2012. A yard waste service line is usually provided all year around, regardless of the actual demand. In 2008, over 3,500 communities were provided a yard waste service line (SWANA, 2008).

2.2. Residential curbside collection routes

An average daily residential collection route varies from 150 to 1300 households, while a commercial route can range between 60 and 400 customers (Kim et al., 2006). Therefore, the collection efficiency of waste collection vehicles with respect to fuel consumption is expected to vary significantly. Larsen et al. (2009) measured the fuel consumption of diesel-fueled waste collection vehicles under 14 different collection patterns in two municipalities in Denmark. The study showed that the diesel consumption ranged between 1.4 and 10.1 L per ton of waste collected, depending on the type of housing and the amount of waste collected per stop (Larsen et al., 2009). The U.S. EPA WARM software estimates diesel collection vehicles fuel consumption at 7.5 L per ton of waste collected (U.S. EPA, 2012a). Fitzgerald et al. (2012) compared the greenhouse gas emissions from dual and single stream collection. The collection fleet fuel consumption decreased by approximately 50% when collection was changed from dual-stream to single-stream collection (Fitzgerald et al., 2012). For two case studies examined, the fuel consumption for dual-stream collection was 20.9 and 23.7 L per metric ton of recyclables collected, compared to 13.7 and 14.8 L per metric ton of recyclables for single-stream (Fitzgerald et al., 2012). Nguyen and Wilson (2010) estimated RCC fuel consumption for normal and co-collection waste collection vehicles at 1.8 L and 1.26 L of diesel per km, respectively, while travelling within the collection areas. The study showed that both vehicles consumed 60% of the total route fuel while actually collecting waste. Additionally, around 5-6 times as much fuel was needed to collect a kg of waste from rural areas compared to urban areas. The study also suggested that reducing the loading time per stop does not significantly reduce fuel consumption.

In comparison of dual-stream with single-stream, dual collection requires two waste collection vehicles or a waste collection vehicle with two compartments (Fitzgerald et al., 2012). In the first scheme, the use of two waste collection vehicles, increases the cost and the environmental impact of recyclables collection. In the second scheme, co-collection waste collection vehicles are designed with two same-size compartments; one for papers, cardboard, magazines, and books, and the other for commingles (aluminum, plastic, and glass). In single-stream, the more efficient use of the truck capacity by combining paper and commingles (aluminum, plastic, and glass) increases the packing density, therefore the waste collection vehicle trip is limited by weight rather than volume (Fitzgerald et al., 2012). In a similar field observation, the paper compartment of dual compartment waste collection vehicle filled more quickly requiring the driver to interrupt the route to empty the vehicle, when the other compartment was not full (personal communication with Josef Grusauskas, 2012); this leads to inefficient use of the truck capacity. Clearly, collection efficiency of waste collection vehicles is highly correlated to the collection scheme.

2.3. Alternative fuel waste collection vehicles

The rising cost of diesel and increasingly stringent regulations are driving the industry to use alternative fuels. Table 2.1 lists possible alternatives to diesel, however only four fuels are commercially available; including diesel, natural gas, biodiesel, in addition to hydraulic-hybrid. Hydrogen and dimethyl ether (DME) fueled waste collection vehicles are still in the research and development phases (FAUN Umwelttechnik, 2011; Arcoumanis et al., 2008).

Table 2.1: Fuel Alternatives for Waste Collection Vehicles

Fuel Category	**Fuel Source**	**Vehicles Availability**	**Source**
Gasoline	Fossil Derived Fuel	Only small waste collection vehicles run on gasoline	U.S. DOE (2012a)
Diesel	Fossil Derived Fuel	The majority of waste collection vehicles run on diesel	U.S. DOE (2012a), Gordon et al. (2003), Rogoff et al. (2009)
Natural Gas	Fossil Derived Fuel or waste degradation (i.e. landfill gas (LFG))	Commercially available as Compressed Natural Gas (CNG) and Liquefied Natural Gas (LNG)	Gordon et al. (2003) NGVAMERICA (2012)
Biodiesel	Biogenic Fuel	Diesel waste collection vehicles can run on biodiesel	Lapuerta et al. (2008)
Hydraulic-hybrid	Fossil Derived Fuel	Commercially available as hybrid vehicles	Hall (2010)
Hydrogen Gas	Fossil Derived Fuel or Water	Available commercially with fuel cell for emptying bins and loading waste (under testing)	FAUN Umwelttechnik, (2011)
Dimethyl Ether	Biogenic Fuel	Not available commercially	Arcoumanis et al. (2008)

In the U.S., the relatively low prices of natural gas have increased industry's interest in natural gas waste collection vehicles (Maimoun et al., 2013). In 2012, waste collection vehicles and transfer vehicles accounted for 11 percent of the total U.S. natural gas vehicles

(NGVAMERICA, 2012). Undoubtedly, the driving factor for the waste industry is fuel cost. In the last three decades, the selection scheme for alternative fuels and energies has changed from a single-criterion cost-based assessment, to a multi-criteria analysis that considers other factors such as environmental, social, operational and political, factors (Pohekar and Ramachandran, 2004; Cavallaro, 2005; Wang et al., 2009; Linkov and Moberg , 2011). Multi-criteria decision analysis (MCDA) problems have been solved using multi-criteria decision making (MCDM) methods that allow decision makers (DMs) to select among alternatives while considering different selection criteria. In the literature, MCDM methods have been used to rank alternative fuel buses for public transportation (Tzeng et al., 2005), alternative transportation fuels (Mohamadabadi et al., 2009), electricity generation alternatives (Cristóbal, 2011), MSW management alternatives (Herva and Roca, 2013), and landfill site selection (Şener et al., 2006). Nevertheless, MCDM methods have not been used to evaluate alternative fuels for waste collection vehicles. Decisions regards alternative fuels for waste collection vehicle are a good candidate for MCDM methods because of the availability of different fuel alternatives and multiple selection criteria that should to be considered by DMs.

2.4. MCDM methods applications in alternative fuel selection

In 1987, Tzeng and Shiau used MCDM methods to evaluate alternative energy conservation strategies for an urban transportation system in Taiwan. The Planning Assistance Through Technical Evaluation of Preference Number (PATTERN) was used to create a system of energy conservation strategies. MCDM methods ELECTRE I&II were applied to rank alternatives under five criteria, including energy conservation, cost, environmental impact, mobility and safety impacts. The study concluded that PATTERN and MCDM methods ELECTRE I&II can be used to solve transportation problems. Poh and Ang (1999) completed a comprehensive study of alternative fuels for land transportation in Singapore. The Analytical Hierarchy Process (AHP) was used to evaluate four possible fuel plans. The most favored fuel plan was to use electric cars. This scenario was different from the most likely future scenario due to the lack of social acceptance of electric cars; therefore an iterative forward and backward AHP planning was used to identify and test policies to achieve the preferred fuel plan. The study concluded that AHP provides a practical decision-making approach for solving the problem.

Tzeng et al. (2005) performed a MCDA of alternative-fuel buses for Taiwan public transportation. AHP was applied to determine the relative weights of evaluation criteria. The Technique for Ordering Preference by Similarity to an Ideal Solution (TOPSIS) and an outranking MCDM method (known as VIKOR) were applied to evaluate the best alternative fuel. Hybrid buses were the best alternative for the Taiwan urban area in both the short and the long term. Mohamadabadi et al. (2009) developed a multi-criteria analysis of the Canadian renewable and non-renewable transportation fuel vehicles. Environmental, economic, and social factors were included in the selection criteria. Preference Ranking Organization Method for Enrichment and Evaluations (PROMTEE) was used as an assessment tool, which can handle both qualitative and quantitative criteria. In the study, two different scenarios were evaluated; (1) higher economic criteria weight, and (2) higher environmental criteria weight. In the first scenario, gasoline was ranked as the leading fuel, while in the second scenario, hybrid vehicles were ranked as the best alternative. Compressed natural gas (CNG) vehicles were ranked last in both scenarios.

2.5. Environmental impacts of residential waste management systems

Waste collection is the first step in any MSW management system. In RCC, waste is collected from single-family households to be transported to a landfill, MRF, composting facility or combustion facility. Globally, up to 95% of MSW collected is landfilled (Diamadopoulos, 1994;

Kurniawan and Chan, 2006). In 2012, about 135 million tons of the U.S. MSW (53.8% of total generation) were discarded in landfills (U.S. EPA, 2014a). Therefore, landfilling is the leading waste management practice in the U.S., followed by recycling and recovering (34.5%), and combustion with energy recovery (11.7%) (USEPA, 2014a). The biodegradation of organics in landfills mainly generates methane (50-60% of volume) and carbon dioxide (40-40%) (Shin et al., 2005; U.S. EPA, 2012b). In 2009, MSW landfills were the third-largest source of human-related methane emissions in the US, accounting for 17% of these emissions (U.S. EPA, 2012c). Therefore, the USEPA requires large landfills to collect landfill gas (LFG) either for beneficial use or flaring (U.S. EPA, 2012c). LFG also consists of hundreds of other compounds at lower concentrations such as oxygen, nitrogen, sulfur compounds, water vapor, and non-methane organic compounds (U.S. EPA, 2000; Shin et al., 2005). The conversion of LFG to vehicular fuel would provide a clean, sustainable and domestic source to fuel local waste collection vehicles (Maimoun et al., 2013). In order to use LFG as an alternative vehicular fuel, LFG should be converted to pipeline quality natural gas, with high BTU content, through the separation of methane from carbon dioxide and other constituents (Hesson, 2008; U.S. EPA, 2000). The second major potential source of pollution from landfills is leachate generation. Landfill leachate consists of liquid generated by the breakdown of waste and the infiltration of precipitation through the landfill (Duggan, 2005). All MSW landfills are required by the federal regulations to install leachate collection and removal system, they are also required to monitor surrounding groundwater (U.S. EPA, 2012c).

The composition of MSW before and after recycling is important to understand the amount of biodegradable organics entering landfills and the effectiveness of the recycling programs (Savage and Demers, 1996; Staley and Barlaz, 2009). Recently, Staley and Barlaz (2009) analyzed eleven statewide discarded MSW streams. Organics (food waste and yard trimming), paper, and plastic components averaged 23.6 ±4.9%, 28.8 ± 6.5%, and 10.6 ±3% of the total generated MSW (Staley and Barlaz, 2009). According to the U.S. EPA (2012a), the national MSW consists of organics (food waste and yard waste), paper and paperboard, plastics, metal, and glass averaging 27.3%, 28.5%, 12.4%, 9%, and 4.6% of the total MSW generated in 2010. The recycling rate of these categories determines the composition of waste entering the landfill (Savage and Demers, 1996).

In 1996, Denison reviewed major North American case studies that compared environmental life-cycle impact of landfilling, incineration and recycling. The reviewed studies compared the MSW management alternatives based on solid waste output, energy use, and pollution released to air and water. All studies concluded that recycling offers substantial life-cycle environmental advantages over virgin production plus either incineration or landfilling. If recycling and waste management activates were considered separately, then virgin material production plus either incineration or landfilling were more beneficial. Morris (2005) compared the life-cycle assessment for curbside recycling versus either landfilling or incineration with energy recovery. The study concluded that recycling of conventionally recoverable materials found in the MSW stream consumes less energy and has lower environmental impacts than landfilling or incineration, even when considering energy recovery for landfilling and incineration. The study also found the energy recovered offsets from LFG or waste combustion are significantly smaller than upstream energy and pollution offsets by recycled material remanufacturing, even after accounting for collection, processing, and transportation of the recovered material.

Villanueva and Wenzel (2007) reviewed a total of nine international life-cycle assessment studies that compared different management options for waste paper. The reviewed studies illustrate the environmental benefits in recycling over incineration or landfill options for paper and cardboard waste. Merrild et al. (2012) showed that there are environmental benefits from recycling paper, glass, steel and aluminum; however, they argued that recycling might not be the right choice if treatment alternative is a waste-to-energy plant with significant energy content materials.

Mendes et al. (2004) performed a life-cycle assessment of the environmental impact of incineration and landfilling of MSW in Sao Paulo City, Brazil. Global warming, acidification, and nutrient enrichment were assessed as environmental impact categories under different scenarios. Overall, the study concluded that landfilling has a higher environmental impact than incineration, whereas the environmental impact of landfilling is slightly reduced by energy recovery. For waste incineration, the reuse of ash resulted in higher environmental impacts compared to ash landfilling due to an increase in energy consumption (Mendes et al., 2004).

2.6. Evaluation criteria of MSW management practices

In the U.S., MSW management practices have been compared with respect to their GHG emissions, pollutant discharges, and cost (Weitz et al., 1999; Weitz et al., 2002). A life-cycle inventory has been used to compare the environmental impact of waste management practices; annual capital and operating costs have also been used as the basis of comparison of waste management practices (Weitz et al., 1999).

The life-cycle of MSW management practices has been incorporated into U.S. and international computer models that compare the environmental impacts of MSW management practices. In the U.S., the Research Triangle Institute and the U.S. EPA have developed the MSW Decision Support Tool (MSW-DST), an online platform used to optimize MSW management based on air emissions, waterborne releases, and cost (Research Triangle Institute, 2004). Using the MSW-DST, Weitz et al. (2002) concluded that advancement in MSW practices between 1974 and 1997 has significantly reduced GHG emissions despite an almost two-fold increase in waste generation rate over this time.

Worldwide studies of MSW management practices have also focused on GHG emissions, cost, ecological footprint, acidification potential, and cost (Consonni et al., 2005; Buttol et al., 2007; Winkler and Bilitewski, 2007; Cherubini et al., 2009). International life-cycle models of MSW management practices are also available. Winkler and Bilitewski (2007) compared six different models developed by research organizations, industry, and government associations. These models, including the MSW-DST, focus on the air pollution emissions, waterborne releases, cost, and, in some cases, soil contaminant releases. The six models, which were used to develop an integrated life-cycle assessment (LCA) of waste management practices to evaluate the air and water emissions were:

- ARES: German model capable of modeling 121 air and 15 water pollutants.
- IWM2: British model capable of modeling 24 air and 27 water pollutants.
- ORWARE: Swedish model capable of modeling 69 air and 68 water pollutants.
- EPIC-CSR: Canadian model capable of modeling 12 air and 5 water pollutants.
- MSW-DST: American model capable of modeling 23 air and 17 water pollutants.
- UMBERTO: German model capable of modeling unlimited number of air and water pollutants.

Winkler and Bilitewski (2007) compared the air and waterborne emissions estimates of these models for their case study. The comparison revealed significant differences in the outputs of these models, potentially due to the assumption of different boundaries for each MSW management life-cycle analysis.

Arena et al. (2004) performed a life-cycle assessment of energy and resource consumption, climate change, acidification, and other emissions potential of three alternative management options (landfilling, recycling, and combustion with energy recovery) for paper and board packaging waste in Italy. The study concluded that material recycling may not be the best environmental option for Italy, while energy recovery from paper is preferred over landfilling and recycling. The study evaluated the water consumption related to landfilling, recycling, and combustion of paper. Landfilling and combustion with energy recovery consumes large quantities of water (more than 50 metric ton of water per ton of paperboards managed). However, the recycling scenario showed very low emissions of pollutants into water (Arena et al., 2004). Arena et al. (2003a) performed a life-cycle assessment of a plastic packaging recycling system. The study also evaluated the water consumption of different plastic packaging recycling scenarios.

Arena et al. (2003b) compared the environmental performance of alternative solid waste management practices for the rest-waste and recycling residue in Italy. The study compared life-cycle emissions, energy consumption, water consumption, greenhouse gases emissions, and acidification potential for three different scenarios that included (1) landfilling, (2) sorting facility which collect ferrous materials and a biological treatment process for the organic fraction, and waste-to-energy for the residue, and (3) combustion. The study accounted for the avoidable direct water consumption as a result of material recovery by sorting and energy recovery by combustion. Moreover, the study accounted for the water consumption for the sorting process and ash conditioning. The study assumed zero water consumption for the waste landfilling process. The study concluded that avoided water consumption from waste incineration by energy production does not offset the water consumption by the waste incineration facility (Arena et al., 2003b).

In the literature, the impacts of some MSW management practices on water resources as waterborne emissions have been evaluated (Weitz et al., 1999; Weitz et al., 2002; Consonni et al., 2005; Winkler and Bilitewski, 2007; Buttol et al., 2007; Cherubini et al., 2009). Also, the direct consumptive uses of water resources have been calculated for a few MSW management practices, i.e. recycling and combustion (Arena et al., 2003a; Arena et al., 2003b; Arena et al., 2004). However, these methods (waterborne emissions and water consumption calculations) do not fully reflect the total effect of MSW management practices on water resources. During the last decade, the WFP methodology (Hoekstra and Hung, 2002; Hoekstra, 2003; Chapagain and Hoekstra, 2007; Hoekstra et al., 2009) has been developed and used to capture both direct and indirect effects of processes, products, entities, industries, energy sources, and countries on water resources (Chapagain et al., 2006; Chapagain and Hoeskstra, 2007; Dominguez -Faus et al., 2009; Gerbens -Leenesa et al., 2009a; Gerbens -Leenesa et al., 2009b), providing a reliable criterion for evaluating water use efficiency.

WFP is a state-of-the-art measure of both the direct and indirect use of fresh water over the entire process life cycle and consists of three components. The blue WFP accounts for the consumption of blue water resources (surface and groundwater). The green WFP refers to consumption of green water resources (rainwater stored in the soil as soil moisture, normally lost through evapotranspiration). The grey WFP is related to water pollution and is defined as the volume of freshwater that is required to dilute pollutants to meet existing water quality standards (Hoekstra et al., 2009).

In this study, a comprehensive assessment of RCC programs will be performed; multiple design factors affecting the efficiency of the U.S. RCC programs will be considered. The study will measure recycling efficiency, GHG emissions, and collection cost of RCC programs as a function of recycling participation rate, collection frequency, and collection system design (dual or single stream). Moreover, MCDM methods will be used to rank fuel alternatives for waste collection vehicles with respect to environmental and financial and criteria. A sensitivity analysis will be conducted to evaluate the robustness of the results to future energy and technology scenarios. Finally, the WFP methodology will be applied to estimate the direct and indirect impacts of common MSW management practices on water resources.

CHAPTER 3: AN ENVIRONMENTAL-ECONOMIC ASSESSMENT OF RESIDENTIAL CURBSIDE COLLECTION PROGRAMS IN FLORIDA

3.1. Introduction

Residential waste collection services provide waste removal from both single family and multi-family dwellings. A single family dwelling is an individual structure with its own lot and is usually serviced by residential curbside collection (RCC), whereas multi-family dwellings are connected structures and are usually provided with dumpsters for waste collection. RCC (the main focus of this study) includes over 8,660 programs throughout the U.S. (Smith, 2012) and serves 71% of the U.S. population (U.S. EPA, 2011a). Collection programs are established by waste management divisions (cities, municipalities, or counties) to provide waste collection and management services for residents. RCC programs usually provide garbage, recyclables, yard waste, and in some cases, food waste collection lines. Typically, such service necessitates a minimum of three weekly collections. These collection services are provided consistently throughout the year for public convenience, although waste generation rates and collection needs vary seasonally, e.g., during holidays and low-growth vegetation seasons (Maimoun et al., 2013).

In the past, populations in the northern part of the US were served weekly by one day of waste collection, whereas the southern part of the US was served weekly by two days of waste collection to minimize odors (Kim et al., 2006). However, RCC programs are faced with rising collection costs due to increasing fuel prices and diversion of waste to recyclable and yard waste lines, providing impetus to switch to once per week or every other week (bi-weekly) waste collection. On the other hand, the main disadvantage of reducing waste collection frequency to weekly or bi-weekly is the health concern associated with leaving food waste in containers for up to two weeks (McLeod and Cherrett, 2008).

In the U.S., the implementation of curbside collection of recyclables increased recycling, diverting reusable materials from the waste stream (U.S. EPA, 2011a). However, customer's convenience plays an important role in the amount of the recovered material. Everett and Peirce (1993) studied the effect of collection frequency, collection day, and containers on material recovery rate for voluntary and mandatory curbside recycling programs. The study concluded that providing containers slightly improved curbside recovery rate for voluntary collection program, but not mandatory programs. On the other hand, increasing recyclables collection frequency had a slightly positive effect on the recovery rate, while collection day had only a slight effect on that. Lave et al. (1999) argued that for most municipal solid waste recycling categories the costs of collection and processing exceeded the avoided disposal fee and revenues from the sales of recyclables.

Recyclables curbside collection can be classified according to the number of collection streams. In the U.S., single-stream (SS) and dual-stream (DS) collection are most common. DS collection requires residents to separate cardboards, papers, and magazines from the rest of recyclable materials using 60-liter (16-gallon) bins, while single stream collection allows residents to mix all recyclable material together using 60-liter (16-gallon) to 240-liter (64-gallon) containers. The number of containers provided for residents varies based on the collection system used and the hauling contract. During the last decade, many communities in the US have switched from DS recyclables collection to SS collection for the ease of operations (Fitzgerald et al., 2012). On average, 14 new SS material recovery facilities (MRFs) have been added every year since 1995 (Berenyi, 2008; Fitzgerald et al., 2012). Fitzgerald et al. (2012) examined the quantities of recycled material at three MRFs and concluded that switching from DS collection to SS generated 50% more recyclables. Jamelske and kipperberg (2006) found that consumers are willing to pay for the combined switch to automated solid waste collection and SS recycling in Madison, Wisconsin. The study presented a positive net benefit from moving to SS recycling with automated collection.

In Europe, Tucker et al. (2001) evaluated the integrated effects of reducing the frequency of curbside collection of newspapers in the UK from once every two weeks to once every four weeks. The study reported a 41% saving in fuel usage, which obviously had environmental benefits as well as cost savings of 60%. However, the net environmental benefits were less than 41% as more residents transported their recycles to collection centers. It was estimated that tonnage recovered suffered a loss of less than 2%, while participation in the curbside collection program dropped by less than 8%. McDonald and Oates (2003) found that the main reasons for non-participation in a curbside recycling scheme of paper within a UK community were lack of insufficient paper and lack of space to store recycling bins. However, the study also reported that more than half of non-participating customers recycle paper using other facilities. The study recommended changing the scheme design (mainly the color of recycling bins), scheme operation and promotion to encourage recycling. In Australia, Gillespie and Bennett (2012) estimated the willingness of households to pay for curbside collection of waste and recyclables. The study observed that respondents had a positive willingness to pay for once every two weeks or once a week collection services, while being less willing to pay for twice a week collection. Understanding the factors affecting recycling behavior is essential to increasing recycling participation (Williams and Cole, 2013). Two trials in England compared the recycling participation associated with changing to SS or DS, while reducing recyclables collection frequency. There was no difference in the recycling participation between SS and DS trials. In comparing DS and SS, Williams and Cole (2013) found that DS collected an average of 5.94 kg/household/week compared to an average of 5.63 kg/ household /week by SS.

The design of RCC programs varies significantly among U.S. areas; major differences are the number of collection lines provided (defined as the number of collection services provided to a resident); the collection frequency of each service line; the type of recycling collection system (DS or SS); the number, type, and volume of garbage and recycling containers; and the fuel used. These variables can significantly affect the recycling efficiency and participation rate of RCC programs. As municipalities try to balance environmental and financial impacts of collection services and customer satisfaction, optimal design of the RCC system will be their first step toward sustainable waste management. Accordingly, this research explores the trade-offs between environmental and economic factors to optimize RCC systems.

In 2012, Florida municipal solid waste (MSW) was generated by single-family dwellings (32% of the total generation), multi-family residences (13%), and commercial entities (55%) (FDEP, 2014a). Approximately, 35% of the total MSW stream was recycled (FDEP, 2014b). Florida state has an ambitious recycling goal of 75% by 2020 (FDEP, 2013), calling for municipalities throughout the state to modify RCC programs as a mean to improve recycling. To increase the recycling efficiency, many municipalities have switched to SS recyclables collection. Moreover, some RCC programs have provided residents with multiple or larger recycling containers to encourage residents to recycle more. At the same time, many collection providers are switching to less frequent garbage collection, due to waste diversion to other service lines (e.g. recyclables and yard waste) and the rising cost of collection. As a result, a variety of program designs were found across the state of Florida, providing a good opportunity to study the effects of the RCC system design on waste generation rates and recycling efficiency. An environmental-economic assessment model was developed and used to estimate the life-cycle greenhouse gas (GHG) emissions and cost of Florida RCC programs using data provided by commercial haulers.

The developed model was used to evaluate the sensitivity of the model outcomes to changing input parameters, in particular, the recycling participating rate (PR_R), and to determine the minimum required PR_R to make curbside recyclables collection environmentally and economically beneficial.

3.2. Methods

Data collection of 112 Florida's RCC programs, serving about four million single-family households, was conducted using municipality websites. Based on the survey, communities were grouped into four sets based on their RCC garbage, yard waste, and recyclables collection design, i.e., frequency of collection and use of dual-stream (DS) or single-stream (SS) recyclables collection system. For this study, communities, haulers, and municipalities in Central Florida area were randomly asked to provide data for this study. The selection of Central Florida area was to ensure the same demographics of population. Only few communities, haulers, cities, or municipalities agreed to provide data. Twenty-five different Floridian communities, serving about half million households, were identified to participate. The rest of this Section will discuss data collection and analysis for the 25 RCC programs, followed by the development of an environmental-economic assessment model.

3.2.1. Hauling Data and Recovered Materials

Each commercial hauler for the 25 identified Florida communities was asked to report the method of collection, collection schedule, number of households served, and the collected tonnage of garbage, recyclables, and yard wastes during years 2009, 2010, 2011 or 2012 (Table A.1 in the Appendix). The composition of recyclables leaving SS and DS MRFs during 2012 was obtained from local facility operators (Table A.2 and Table A.3 in the Appendix). The U.S. EPA Waste Reduction Model (WARM) version 13 (U.S. EPA, 2014b) was used then to estimate GHG emission offsets resulting from recycling through RCC programs. The contamination rate (the portion of recyclables that was contaminated during collection and could not be recycled, i.e. the waste residue) was evaluated by analyzing the composition of materials leaving DS and SS MRFs and validated by hand-sorting of individual collection vehicle contents by commercial haulers. The waste residue reported by the SS MRF was 9.07% compared to a 10.40% reported by the DS MRF. Therefore, for the purpose of this study, 10% of all collected DS and SS recyclables was assumed to be later diverted to landfills.

3.2.2. *Analysis of Waste Generation Characteristics*

The total household waste generated was defined as the sum of garbage and recyclables, excluding yard waste. The generation rate of total household waste was calculated using Equation 3.1.

$$GR_T = \frac{(W_G + W_R) \times 1000}{N_T \times 365} \tag{3.1}$$

where:

GR_T: Generation rate of total household waste (Kg per served household per day)

N_T: Maximum number of households served by collection contract

W_G: Annual weight of garbage collected from N_T customers (Metric Ton (MT) per year)

W_R: Annual weight of recyclables collected from N_T customers (MT per year)

Recycling Percentage (RP) was calculated as the percent of GR_T that was recycled, as shown in Equation 3.2.

$$RP\ (\%) = \frac{W_R}{W_R + W_G}\ x\ 100\% \tag{3.2}$$

Statistical analysis of garbage, recyclables and total waste generation rates was performed using Minitab 16. Because of the small sample sizes, data were not normally distributed. Therefore, the Mann-Whitney U test (Mann and Whitney, 1947) was used to compare the equality of generation rate medians associated with SS and DS recyclable systems, and two and one-day garbage collection data, using a 95% significance level to interpret the results. The non-parametric Mann-Whitney U test is used when parametric z or t tests cannot be used; because the assumptions related to level of measurement, sample size, normality or equality of the variance are not valid (Butler, 1985).

3.2.3. The Environmental-Economic Assessment Model

An environmental-economic assessment model was developed and used to estimate the GHG emissions and cost of Florida RCC programs as a function of recycling participation rate (PR_R, percent of households' participating in curbside recycling). A sensitivity analysis of the results was performed to evaluate the effect of input parameters on model outputs.

3.2.3.1. Waste Generation Rate as a Function of PR_R

The generation rate of recyclables per participating household (GR_R, kg per participating household per day) was calculated using PR_R as shown in Equation 3.3.

$$GR_R = \frac{W_R \times 1000}{PR_R \times 365 \times N_T} \quad (3.3)$$

In order to calculate the average garbage and recyclables generation rate per household served by collection contract (kg per served household per day), it was assumed that the reported collected tonnage was generated by the total number of households served by collection contract. A statistical analysis was used to test the research hypothesis that Florida's households generate similar quantity of total waste regardless of the RCC program characteristics.

Recyclables collection diverts recyclables from the garbage collection line; the higher the system participation rate and recycling percentage, the less garbage is collected. In 2012, the average recycling participating rate reported in Florida curbside collection programs was 67% (FDEP, 2014c). The average recycling participation rate varied significantly across Florida, thus this study was designed to understand the impact of recycling participation rate on the environmental and economic performance of RCC programs. In this study, garbage participation rate (PR_G) was assumed to be 100%, based on the haulers' input. PR_R was reported to be 70% by only four of the 25 Florida communities; this value, 70%, was used to analyze the environmental and economic impacts for all 25 communities. The garbage generation rate can be calculated as a function of the PR_R, as shown in Equation 3.4, to determine the impact of this parameter on the environmental and economic performance of RCC programs.

$$GR_G = \frac{W_G \times 1000}{PR_G \times 365 \times N_T} = \frac{GR_T - (PR_R \times GR_R)}{PR_G} \qquad (3.4)$$

3.2.3.2. <u>Households Served per Collection Trip as a Function of PR_R</u>

During each collection trip, a waste collection vehicle starts at the garage and then travels to the collection site where it stops at participating households. At the end of the collection trip, the vehicle transports the collected material to the post-collection facility (e.g., a landfill, transfer station, waste-to-energy facility, or MRF). Then, the waste collection vehicle travels empty from the post-collection facility back to the garage. Time and fuel use for curbside waste collection can be considerably different depending on the housing density along the collection route, however it was estimated that the fuel consumption during waste collection accounts for more than 60% of the total daily fuel use (Nguyen and Wilson, 2010). Because the focus of this study was on waste collection activities that consume most of the fuel and are most impacted by PR_R, this analysis only reflects emissions and costs for a single collection trip. It was assumed that the characteristics (distance and time) for travel between the garage and collection site, between the collection site and post-collection facility, and between the post-collection facility and garage, are constant for all the tested RCC systems, as well as break times and unloading time at the post-collection facility.

Default values for model variables are given in Table 3.1. For a single trip, the number of households that can be served was constrained by the truck legal weight limit - difference between the gross vehicle weight rating and curb weight -(C, MT) for garbage and yard waste, truck volume (V, m^3) or driver daily hours (T, hours) for recyclables. The maximum number of households that can be served for garbage collection during one trip can be calculated based on truck's legal weight and generation rates of garbage using Equation 3.5.

$$N_{G^*} = \frac{C \times 1000}{\frac{7days}{week} \times GR_G} \tag{3.5}$$

where:

N_{G^*}: Maximum number of households that can be served for garbage collection during a single collection trip.

In case of two days of garbage collection per week, it was assumed that two-thirds of the weekly garbage generation will be collected on the first day, while the rest will be collected on the second day.

Table 3.1: The values of the environmental-economic assessment model's input variables.

Model Inputs	Symbol	Default Value	Unit	Justification/Reference
Distance between household	D_{HH}	22.3 (±14.6)	m (meters)	Distance between households based on a random 20 Florid household's sample.
Travel speed between households	S_{HH}	10	Km/h (kilometers per hour)	Assumed travel speed
Time to collect garbage per household	$T_{1(G)}$	8.74	S (seconds)	Curtis and Dumas (2000)
Time to collect DS recyclables per household	$T_{1(DS)}$	27	S	Curtis and Dumas (2000)
Time to collect SS recyclables per household	$T_{1(SS)}$	9	S	Curtis and Dumas (2000)
Truck legal weight	C	10.4	MT (metric tons)	Commercial haulers' specifications
Truck volume	V	24.5	m^3	Commercial haulers' specifications
Driver daily hours	T_{max}	10.5	h (hours)	Commercial haulers' specifications
Lunch and Break	L&B	60	Min (minutes)	Curtis and Dumas (2000)
Vehicle driving range	R_{max}	240	Km (Kilometer s)	Commercial haulers' specifications
Distance from garage to start collection (Garbage and Recyclables)	D_{GA}	19	Km	Curtis and Dumas (2000)
Time from garage to start collection (Garbage and Recyclables)	T_{GA}	20	Min	Curtis and Dumas (2000)
Distance from post-collection facility to garage (Garbage and Recyclables)	D_{FG}	19	Km	Curtis and Dumas (2000)
Travel time from post-collection facility to garage (Garbage and Recyclables)	T_{FG}	20	Min	Curtis and Dumas (2000)
Distance from collection site to post-collection facility (Garbage)	$D_{F(G)}$	35	Km	Curtis and Dumas (2000)
Travel time from collection site to post-collection facility (Garbage)	T_F	44	Min	Curtis and Dumas (2000)
Distance from collection site to post-collection facility (Recyclables)	$D_{F(R)}$	35 (DS); 37 (SS)	Km	Curtis and Dumas (2000)
Travel time from collection site to post-collection facility (Recyclables)	$T_{F(R)}$	46 (DS); 44 (SS)	Min	Curtis and Dumas (2000)

In case of one day of recyclables collection per week, the maximum number of households that can be served for recyclables during one trip (N_{R^*}) can be calculated based on V, specific weight (SW, Kg/m^3), GR_R and PR_R using Equation 3.6. Based on field data from the haulers, the SW of recyclables was set to 90 and 130 Kg/m^3 for collection without and with compaction, respectively. Equation 3.6 was used to estimate the number of households that can be served for recyclables collection at different PR_R, while using DS or SS collection, with or without compaction.

$$N_{R^*} = \frac{V \times SW}{\frac{7\ days}{week} \times GR_R \times PR_R} \tag{3.6}$$

3.2.3.3. <u>Collection Speed as a Function of PR_R</u>

For a single daily trip, it was assumed that a waste collection vehicle will not exceed the default driver daily hours (T_{max} = 10.5 h) or the driving range (R_{max} = 240 km). In the case of low waste generation or participation rate, the waste collection vehicle will have to stop collecting and head back to the post-collection facility due to either driver or driving range constraint and the truck will reach the post-collection facility less than full. An increase in PR_R will result in greater amount of recycled material; however, this will be accompanied by increased collection time for the same total collection distance and subsequently a reduced average speed. The average speed associated with waste collection was calculated by dividing the total distance travelled (distance between consecutive houses multiplied with number of houses served), by total time (estimated as sum of time traveling between consecutive houses and collection time at stops). The average collection speed of recyclables (S_R, km/h) and garbage (S_G, km/h) were calculated using Equations 3.7 and 3.8. The time to collect recyclables per participating household (T_1) depends on the type of collection system, i.e., DS ($T_{1(DS)}$) or SS ($T_{1(SS)}$).

$$S_R = \frac{D_{HH} \times (N_{R^*}-1)}{(N_{R^*}-1)\times\left[\frac{D_{HH}}{1000\times S_{HH}}\right]+PR_R\times(N_{R^*})\times\left[T_{1(DS)}\ or\ T_{1(SS)}\right]} \tag{3.7}$$

$$S_G = \frac{D_{HH}\times(N_{G^*}-1)}{(N_{G^*}-1)\times\left[\frac{D_{HH}}{1000\times S_{HH}}\right]+(N_{G^*})\times\left[T_{1(G)}\right]} \tag{3.8}$$

where:

D_{HH}: Distance between households (m)

S_{HH}: Travel speed between households (km/h)

3.2.3.4. Collection GHG Emissions

Garbage collection GHG emissions (Kg CO_{2eq} per MT of garbage) consist of the summation of collection, garage-to-collection site, collection site-to-post-collection facility, and post-collection facility-to-garage emissions, divided by the collected garbage tonnage. The emission factor (kg CO_{2eq} per km travel) associated with each driving mode was estimated using the average speed calculated based on default driving distance and time listed in Table 3.1. In this study, the fuel mileage of garbage, recyclables, and yard waste collection vehicles was obtained from commercial haulers for different travel speeds. According to GREET (2012), the lower heating value of one liter of diesel is 36,090 kilojoules (kj), and the well-to-wheel GHG emissions (summation of well-to-pump and pump-to-wheel emissions) associated with each kj is equal to 0.095 grams of carbon dioxide equivalent (CO_{2Eq}) (U.S. DOE, 2012b). Therefore, 3,430 grams of CO_{2eq} are emitted per liter of diesel burned. The average garbage collection speed was estimated using Equation 3.8 and the variable values given in Table 3.1. The same approach was used to calculate the GHG emissions associated with recyclables collection (Kg CO_{2eq} per MT of recyclables). However, for recyclables, the collection emissions were offset by -2.2 MTCO2eq per MT of recyclables collected using DS or SS collection system. Emission offsets were calculated using WARM version 13 and the recyclables composition leaving SS and DS MRFs provided in Table A.2 and Table A.3 in the Appendix. This estimate accounted for each material loses during remanufacturing as specified by WARM. For this study, additional emissions credits associated with diverting recyclables from landfills or other traditional MSW management facilities were not added to the benefits of recycling. The GHG emissions of the total collected household waste were the summation of the GHG emissions of garbage collection and the net GHG emissions of recyclables collection as shown in Equation 3.9.

$$CE_T = (1 - RP) \times CE_G + RP \times (CE_R - O_R) \quad (3.9)$$

where:

CE_T: Net collection GHG emissions (Kg CO_{2eq} per MT of total household waste generated)

CE_G: Garbage collection emissions (Kg CO_{2eq} per MT of garbage collected per trip)

CE_R: Recyclables collection emissions (Kg CO_{2eq} per MT of recyclables collected per trip)

O_R: Recyclables emissions offset (Kg CO_{2eq} per MT of recyclable collected per trip)

3.2.3.5. <u>Collection Cost</u>

Collection cost is a function of the initial (capital) costs of vehicle acquirement, fuel mileage of waste collection vehicles, driving routes, truck maintenance costs, driver hourly rates, and overhead management costs. In this study, the overhead management and vehicle initial costs were excluded because they are independent of the driving hours and distances related to RCC system design. The collection cost per trip was measured as a function of driving hours and driving distances, fuel cost, and maintenance and labor cost. In Florida, the avoided costs from recyclables diversion were \$60-80 per ton for waste-to-energy, and \$40 per ton for landfilling. The processing cost of recyclables at a MRF can also be significant. Dubanowitz (2000) estimated that the processing cost of recyclable at \$127 per ton of material diverted. The average selling price of recyclables varies significantly and currently average \$100 per MT ton. For this study, net revenues (generated by selling recyclables and avoided disposal cost, and adding MRF cost) were subtracted from the collection cost. Three net revenues scenarios were considered: \$50, \$100, \$150 per MT of recyclables. The net collection cost of recyclables was calculated for the RCC programs, varying PR_R, fuel cost, and recyclables revenues at constant maintenance cost and labor wages, because maintenance cost and labor wages are more stable than fuel cost and recyclables revenues. Collection vehicle maintenance cost was reported by commercial haulers at $8.5 per hour of truck operation, while hourly labor wage for haulers was assumed to be $20 per hour.

3.3. Results

The online survey found that 58% of Florida RCC programs utilize SS recycling system and 38% utilize DS recycling system, whereas 4% do not provide any curbside recycling program. Weekly collection schedules were found to vary considerably, with 49% of RCC programs providing two days of garbage (G), one day of recyclables (R), and one day of yard waste collection (YS) [represented by (2G, 1R, 1YW)] and 29% providing one day of garbage, one day of recyclables and one day of yard waste collection (1G, 1R, 1YW). The remaining programs used a variety of collection system designs, but for the most part provided one or two days of garbage collection, no or every-other week recyclables collection, and every-other week yard waste collection. The selected 25 Florida RCC systems reflected the survey findings and were placed into four categories, representing Florida's most common RCC programs, based on their collection schedule and recyclables collection system as follows:

Group A: 2G, 1R, 1YW-DS Collection (16 communities)

Group B: 1G, 1R, 1YW-DS Collection (3 communities)

Group C: 2G, 1R, 1YW-SS Collection (4 communities)

Group D: 1G, 1R, 1YW-SS Collection (2 communities).

Garbage containers ranged in size from 79 to 360 liters (21 to 96 gallons), while recycling containers were either 61-liter (16-gallon) bins or 240 to 340-liter (64 to 90-gallon) toters. In general, toters were only used with the SS recyclables collection system, while bins were used mainly with the DS system, but in few cases, they were used with the SS recyclables collection system.

3.2.1. Waste Generation Characteristics of RCC programs

The program design, household count, and the reported tonnage of the 25 studied Florida communities are provided in Table A.1 in the Appendix**.** The median garbage generation rate of SS programs was slightly less than DS programs. However, the difference was not statistically significant (p=0.117) (Figure 3.1a). Similarly, the difference between the median garbage generation rate of two days versus one day of garbage pickup was insignificant (p=0.642). Overall, the mean garbage generation rates for SS and DS recycling programs were 2.32 (±0.71) and 2.69 (±0.47) kg per household per day, respectively.

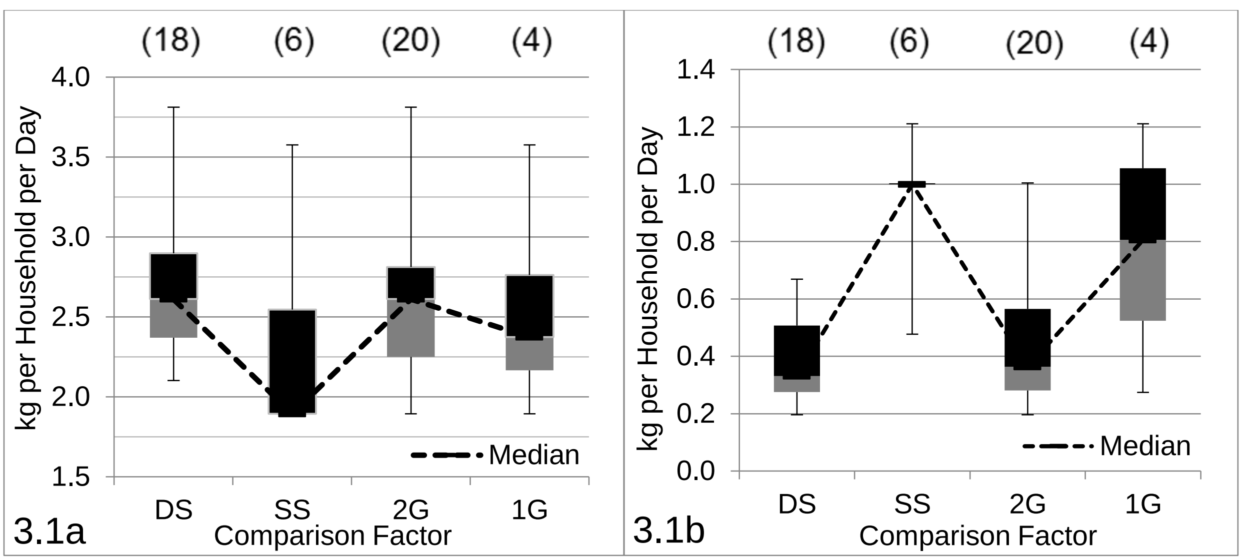

Figure 3.1: Garbage and recyclables generation rates of dual-stream (DS), single-stream (SS), two- day garbage collection (2G), and 1-day garbage collection (1G) RCC programs. (Box-whisker plots of (a) garbage and (b) recyclables generation rates as calculated for program designs, where median values are indicated by the gray-black color interface, box borders denoted 50% interquartile range and whiskers denote data set range. The sample size of each group is given in parentheses.)

In comparing recyclables generation rates, programs implementing SS collection had a significantly higher recyclables generation rate compared to DS programs (p=0.0012) (Figure 3.1b). The recyclables generation rate was not significantly different when RCC programs were stratified by days of garbage collection (two versus one day of garbage collection, p=0.163). The mean recyclables generation rates for 2G, 1R, 1YW-DS;1G,1R,1YW-DS; 2G,1R,1YW-SS; and 1G,1R,1YW-SS programs were 0.37 (±0.14); 0.44 (±0.24); 0.87 (±0.26); and 1.11 (±0.15) kg per household per day, respectively. Overall, the mean recyclables generation rates were 0.38 (±0.15) and 0.95 (±0.25) kg per household per day for DS and SS, respectively.

The total household waste generation rates are shown in Figure 3.2a. The median total household waste collected from DS and SS programs were not statistically different (p =0.973). Similarly, difference for total waste generation median rates of two days versus one day of garbage collection (p=0.938) was statistically the same. For the 25 studied communities, the overall mean total household waste was 3.11 (±0.56) kg per household per day, while the mean recycling efficiencies were 0.3 (±0.08) and 0.13 (±0.04) for SS and DS recycling programs, respectively. These results support the research hypothesis that, on average, Florida households generate similar quantities of waste (garbage plus recyclables), and the more efficient the recycling system, the less garbage collected.

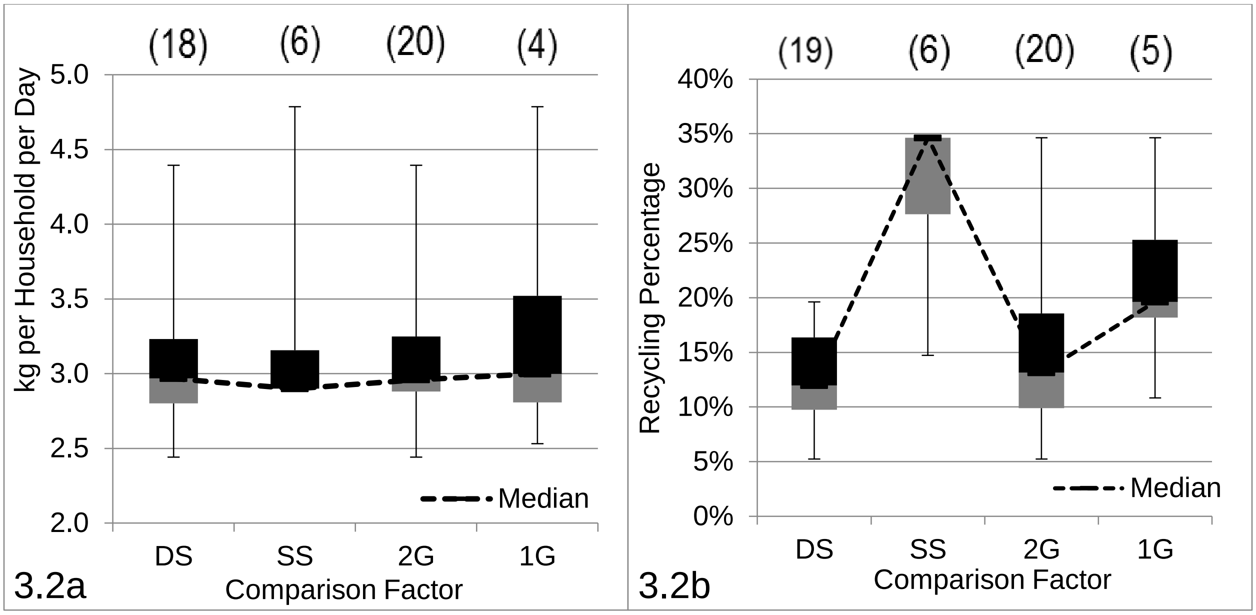

Figure 3.2: Total household waste and recycling percentage of dual-stream (DS), single-stream (SS), 2-day garbage collection (2G), and 1-day of garbage collection (1G) RCC programs. (Box-whisker plots of (a) household total waste and (b) recycling percentage as calculated for program designs, where median values are indicated by the gray-black color interface, box borders denoted 50% interquartile range and whiskers denote data set range. The sample size of each group is given in parentheses.)

The mean recycling percentages for programs 2G,1R,1YW-DS; 1G,1R,1YW-DS; 2G,1R,1YW-SS; and 1G,1R,1YW-SS were 12% (±4%); 16% (±5%), 30% (±10%), and 30% (±10%), respectively. Programs implementing SS collection exhibiting significantly higher recycling percentage in comparison with DS programs (p= 0.001). The recycling percentage was not significantly different when RCC programs were stratified by days of garbage collection (two days versus one of garbage collection, p=0.118).

Recycling percentage ranged 5-20% for DS, and 15-35% for SS. The recycling percentage reported by SS (which serve more than 50% of Florida RCC programs) is close to Florida overall recycling average (35%) in 2012. In comparing DS and SS, the number of bins (DS system) provided for residents varies based on the collection system used and the hauling contract. In general, residents are not willing to use more than two bins due to space limitation (Personal

Communication with Alan Morrison, 2012). It was observed that any recyclables placed outside bins was usually discarded as garbage. Moreover, SS provides bigger containers that does not require residents to cut cardboard boxes (in most cases), thus provides move convenient recycling.

3.2.2. Fuel Consumption of Diesel-fueled Waste Collection Vehicles

The fuel consumption and the associated average speed for typical garbage, recyclable and yard waste collection vehicles, which is linked to approximately 600 waste collection routes in Central Florida, was obtained from commercial haulers. In another study, Farzaneh et al. (2009) reported the fuel consumption of waste collection vehicles for 12 different average speeds. The fuel consumption of waste collection obtained from commercial haulers and Farzaneh et al. (2009) was plotted as a function of the average collection speed as shown in Figure 3.3.

Maimoun et al. (2013) modeled the fuel consumption as a function of the average speed using the U.S. EPA Motor Vehicle Emission Simulator (MOVES) 2010a software (U.S. EPA, 2011c). As shown in Figure 3.3, MOVES underestimates the fuel consumption for the average collection speed of 7 to 25 km/h; this is a result of the numerous driving cycles that can be characterized by the same average speed, as well as vehicle age, engine size, and weight. Overall, the fuel mileage of waste collection vehicles increased from 0.2 and 1.9 Km per liter of diesel consumed as the average collection speed increased from 2 to 25 Km per hour.

After 25 km/h, the fuel mileage of waste collection vehicles increased more consistently with MOVES. The fuel mileage increased slightly from 1.9 to 2.0 km per liter of diesel as the average speed increased 25 to 30 km/h. After 30 km/h according to MOVES (not illustrated by the figure due to the limited field data), the fuel mileage continued to increase slightly to reach 2.6 km per liter of diesel at 60 km/h, reflecting highway driving. Next, field measurements (under 25 km/h) and MOVES estimates (above 25 km/h) of fuel consumption were used to estimate the Florida RCC programs' GHG emissions as illustrated in Section 3.2.3.4.

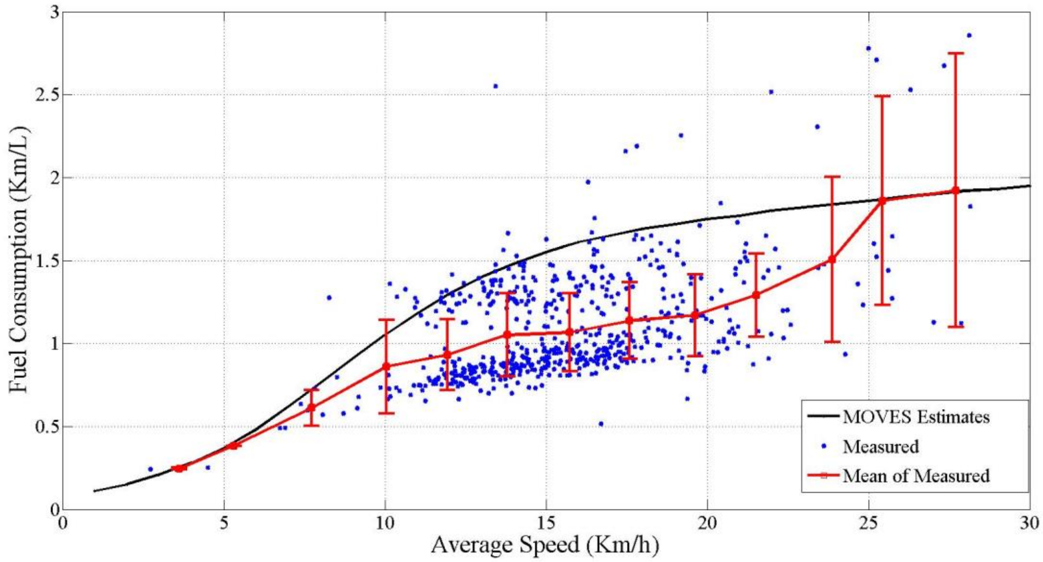

Figure 3.3: Fuel mileage of diesel-fueled waste collection vehicles as a function of average vehicle speed. (The "mean of measured" represents the mean fuel mileage, for diesel-fueled waste collection, measured by commercial haulers (600 data points) and Farzaneh et al. (2009) (12 data points). Whickers denote one standard deviation. The average fuel mileage reported by Maimoun et al. (2013) using the U.S. EPA MOVES 2010a software is represented by the black curve.)

3.2.3. Florida RCC Programs' GHG Emissions

3.3.3.1 Garbage Collection GHG Emissions

As implied by Equation 3.5, customers' participation in recycling diverts recyclables from the total household waste, generating less garbage. On the other hand, non-participating customers dispose recyclables in the garbage collection line and generate more garbage. Thus, as PR_R increases, the number of households that can be served for garbage collection by one vehicle per trip increases.

Figure 3.4 illustrates the maximum number of households (N_{G*}) that can be served for garbage collection by one vehicle per trip as a function of PR_R; the daily limit represents the hypothetical maximum number of household that can be served in 10.5 hours, including breaks.

The number of households served per trip and the associated PR_R were used to calculate the average garbage collection speed (S_G) using Equation 3.8.

The fuel mileage was obtained from Figure 3.3 and was used to estimate the GHG emissions associated with garbage collection (kg CO_{2eq} per MT of garbage) as described in Section 3.2.3.4. As PR_R increases, the number of households served per trip increases; thus the GHG emissions associated with garbage collection (kg CO_{2eq} per MT garbage) increases, as a truck travels and stops more.

The garbage collection's GHG emissions was found to increase from 20 to 30 Kg CO_{2eq} per MT of garbage, for programs with one day of garbage collection as PR_R increased from 0 to 100%. For programs providing two days of garbage collection, the GHG emissions increased from 30 to 45 kg CO_{2eq} per MT of garbage as PR_R increased from 0 to 100%.

In comparison, using the collection model, developed by Curtis and Dumas (2000) and has been incorporated into the US municipal solid waste decision support tool (MSW-DST), the GHG emissions associated with curbside collection of garbage were estimated to be 28.6 CO_{2eq} per MT of garbage. The range observed in this study was the result of accounting for different collection frequencies, recycling generation rates, and PR_R. In another study in Denmark that supports this study findings, Larsen et al. (2009) observed a considerable variation in fuel consumption, and thus the GHG emissions associated with different collection schemes, ranging from 4.8 and 35 kg CO_{2eq} per MT of waste. The GHG emissions associated with single-family waste collection in urban areas, was estimated to be between 11.4 and 12.4 Kg CO_{2eq} per MT of waste, while the GHG emissions associated with rural waste collection was between 22 and 35 kg CO_{2eq} per MT of waste as trucks travel more to collect waste (Larsen et al., 2009). The variances could be linked to the difference in collection schemes, routes, vehicle, and generation rates between the U.S. and Denmark.

Garbage collection emissions were calculated as Kg CO_{2eq} per MT of garbage; however, this analysis cannot be used to compare RCC programs at different PR_R. Emissions should be adjusted to account for the reduction in garbage collection as PR_R increases (Equation 3.9). As PR_R increases, collected garbage decreases, and garbage collection emissions decline by the change in garbage fraction in the total waste stream. Figure 3.5 illustrates garbage collection emissions as Kg CO_{2eq} per MT of total waste. The emission gap between programs 2G, 1R, 1YW and 1G, 1R, 1YW represents the emissions associated with the second day of garbage collection service, resulting in a 50% increase in GHG emissions at PR_R=0%, compared to a 60% and 80% increase in GHG emissions at PR_R=100% for the DS and SS programs, respectively. Collection of less garbage by SS programs allows garbage trucks to serve more households per trip. However for two day per week garbage collection, the second day of garbage collection provided by SS programs was constrained by daily hours at a PR_R of 40% or higher (Figure 3.4). Additionally, the RE of programs using 1G, 1R, 1YW-SS is slightly higher than programs using 2G, 1R, 1YW-SS; therefore, at 100% PR_R, an extra day of garbage collection resulted in an 80% increase in GHG emissions when using SS compared to one day garbage collection (Figure 3.5). As PR_R increased, the emissions associated with programs serviced with SS decreased more than DS programs, due to the effectiveness of the SS system in diverting more waste to recycling.

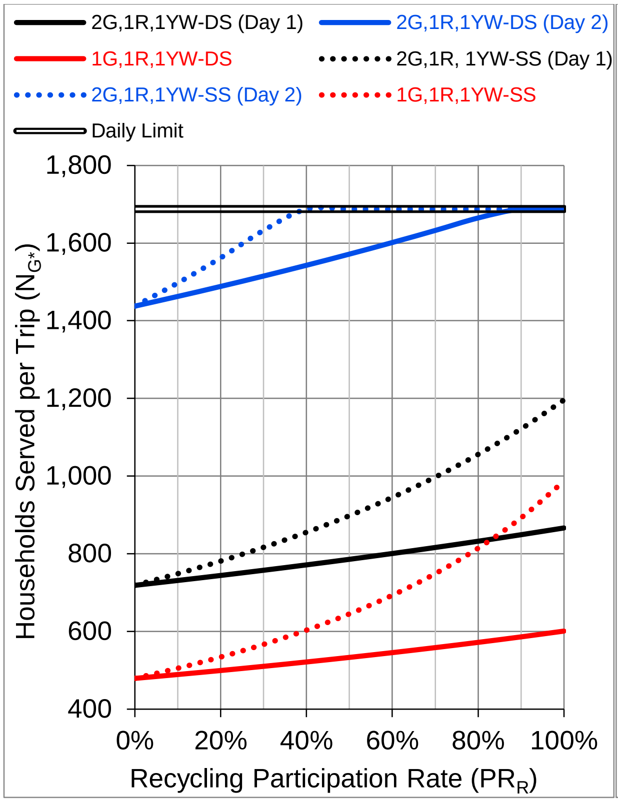

Figure 3.4: The number of households (N_{G*}) that can be served for garbage collection per vehicle per trip.

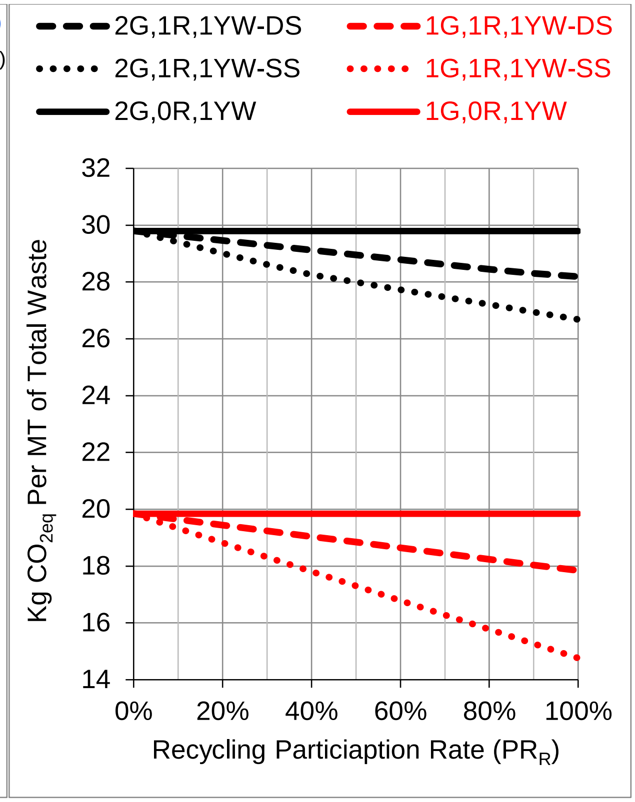

Figure 3.5: GHG emissions during garbage collection as a function of PR_R (kg CO_{2eq} per MT total waste).

3.3.3.2 Recyclable Collection GHG Emissions

Figure 3.5 and 3.6b illustrate the number of households that can be served for recyclable collection by each vehicle per trip based on Equation 3.6. As PR_R increases, the number of dwellings served per trip decreases due to more recyclables pickups. Compaction of recyclables enables serving more households per vehicle per trip, although the quality of recyclables may be reduced. The daily limit represents the hypothetical maximum number of households that can be served within 10.5 hours, including time devoted to non-collection activates. SS programs generate more recyclables per dwelling than DS; thus less households can be served per trip compared to DS. The collection of recyclables without compaction limits the number of households that can be served per trip, while a longer collection time ($T_{1(DS)}$) per stop associated with DS collection can also limit the number of dwellings that can be served per trip, i.e. the number of households served per trip using DS recyclables collection system was limited by the drivers daily hours for any PR_R below 30% and 80% for collection without and with compaction, respectively.

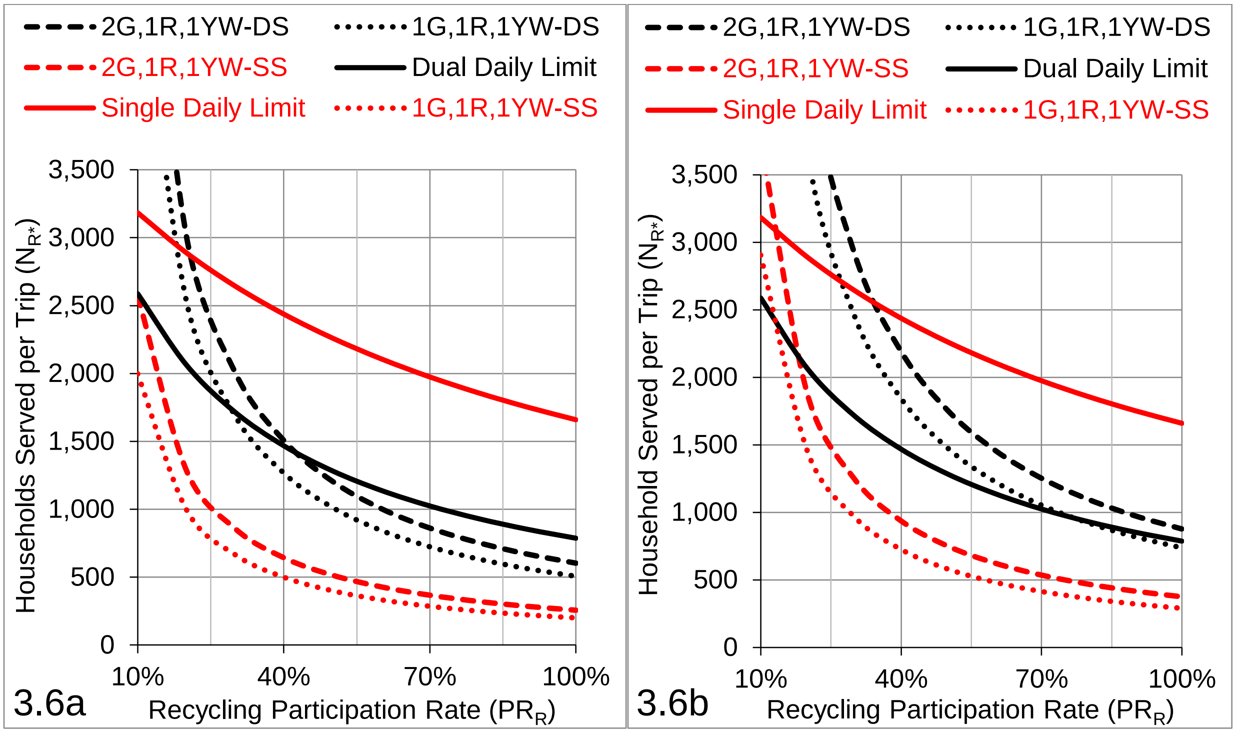

Figure 3.6: The number of households that can be served for recyclables per vehicle per trip, (a) without compaction, (b) with compaction for each program design. The daily limit represents the hypothetical maximum number of households that can be served in one day (10.5 hours including breaks).

The number of household served per trip (N_{R^*}) and the associated PR_R were used to calculate the average collection speed (S_R) using Equation 3.7. The fuel mileage was obtained from Figure 3.3 and was used to estimate the GHG emissions associated with recyclables collection (kg CO_{2eq} per MT recyclables) as described in Section 3.2.3. Although, the average collection speed of recyclables decreases as PR_R increases; it was observed that the GHG emissions associated with recyclables collection (kg CO_{2eq} per MT recyclables) decreases, as a truck travels less to collect the same amount of recyclables. In this study, SS recyclables collection GHG emissions decreases from 155 to 52 kg CO_{2eq} per MT of recyclables as PR_R increases from 10% to 100%, whereas a decline from 480 to 125 Kg CO_{2eq} per MT recyclables was observed for DS collection as PR_R increases from 10% to 100%. SS collection systems provides faster time to collect recyclables (9 seconds per stop) than DS (27 seconds). Therefore, more households can be served and the fuel consumption drops as the average speed of collection is higher. The average collection speed of SS programs was between 4-9 km/h, compared to 2-7 km/h for DS programs. The GHG emissions associated with SS and DS recyclables collection were 101 and 144 kg CO_{2eq} per MT recyclables, respectively (Curtis and Dumas, 2000). In another study, Fitzgerald et al. (2012) reported the GHG emissions associated with recyclables collection at 55 and 77 kg CO_{2eq} per MT of recyclables using of SS and DS, respectively. The results presented here are consistent with literature ranges; this study also found relatively higher GHG collection emissions associated with SS collection compared to DS. The wide range for collection emissions observed in this study demonstrates the significance of considering PR_R in evaluating the environmental impact of recyclables collection.

Recyclables collection emissions were calculated as Kg CO_{2eq} per MT of recyclables; however, this analysis cannot be used to compare RCC programs at different PR_R. Emissions have to be adjusted to account for the increase in recyclables collection as PR_R increases (Equation 3.9). As a result of increase in PR_R, collected recyclables increases, and recyclables collection emissions increase by the fraction of recyclables in the total waste stream. Figure 3.7 illustrates recyclables collection as kg CO_{2eq} per MT of total waste. As PR_R increases, GHG emissions per MT total waste associated with recyclables collection increases.

At any PR_R, GHG emissions from SS recyclables collection systems with compaction are less than DS collection systems, even though SS programs are associated with higher recyclables' generation rate and RE. On the other hand, collection without compaction has higher emissions as less recyclables are collected per trip. The collection emissions of recyclables without compaction for 1G, 1R, 1YW-SS exceed emissions of all DS programs' recyclables' emissions for any PR_R higher than 25%. In case of 2G, 1R, 1YW-SS without compaction, recyclables collection emissions exceed emissions of all DS recyclables collection with compaction for any PR_R above 85%.

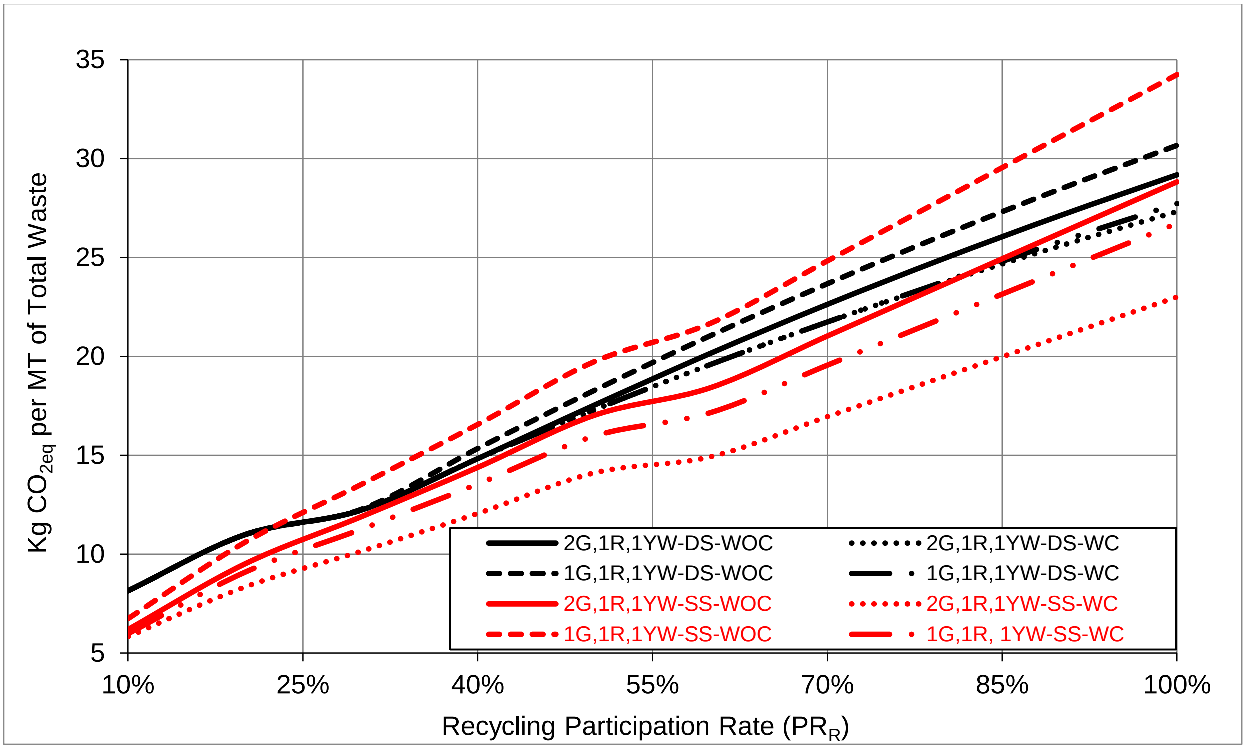

Figure 3.7: Recyclables collection line's GHG emissions. (For each program, emissions were calculated for recyclables collection using SS or DS collection system with compaction (WC) or without compaction (WOC).)

3.3.3.3 Total Waste Collection GHG Emissions

The GHG emissions of the garbage collection line were added to the recyclables collection line to estimate the total collection emissions associated with each program (Figure 3.8a). When PR_R was low, the effect of having a second day of garbage collection was accompanied by a 1.4-fold increase in emissions over programs with one day of garbage collection. An increase in PR_R increased waste diversion, reducing garbage collection emissions while increasing recyclables' collection emissions. The collection of household waste without curbside recycling (2G, 0R, 1YW and 1G, 0R, 1YW), as shown in Figure 3.8a, had relatively low emissions (30 and 19 kg CO_{2eq} per MT of total waste, respectively); however, the quality and cost of recovering recyclables from the mixed waste stream is a concern.

At PR_R=70%, the GHG emissions associated with the four collection programs are estimated to be between 36 and 51 kg CO_{2eq} per MT of total household waste, depending on the garbage collection frequency, recyclables collection system (DS or SS), and recyclables compaction. RCC programs implementing SS recyclables collection with compaction have lower emissions than DS programs. When recyclables offsets were considered (Figure 3.8b), the GHG emissions associated with programs using SS were -760 to -570, compared to -270 to -210 kg CO_{2eq} per MT of total waste for DS programs. In any case, collection emissions were negligible when compared to the benefits of recycling offsets. However, the significance given to collection emissions is urban pollution as the bulk of the emissions are considered tail-pipe emissions.

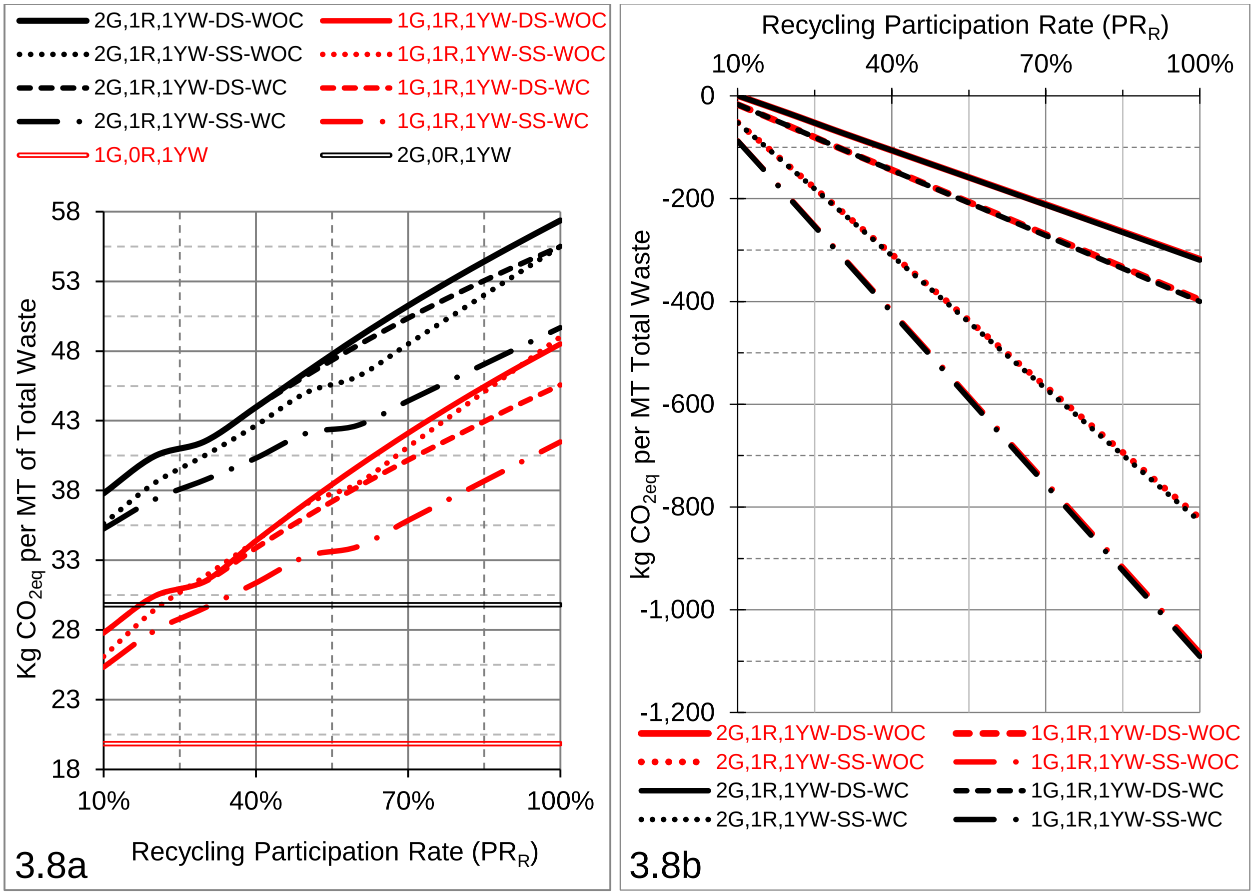

Figure 3.8: Florida RCC programs' total waste collection's GHG emissions, (a) Total waste collection's GHG emissions, (b) Net GHG emissions. GHG emissions were estimated for different RCC system designs as Kg CO_{2eq} per metric ton of total waste (garbage and recyclables) collected. For each program, emissions were evaluated for recyclables collection using SS or DS collection system with compaction (WC) or without compaction (WOC).

3.2.4. Collection Cost of RCC programs

As PR_R increases, the number of households served for garbage collection per trip increases, as a result the fuel consumption (liters of diesel per MT of garbage) and collection time (hours per MT of garbage) increases. The fuel consumption associated with one day of garbage collection increases from 7.2 to 10 L per MT of garbage as PR_R increases from 0 to 100%. On the other hand, programs providing two days of garbage collection had fuel consumption increases from 10 to 15 L per MT of garbage as PR_R increases from 0 to 100%. Larsen et al. (2009) also observed a considerable variation in the fuel consumed for different collection schemes in Denmark, ranging from 1.4–10.1 L diesel per ton of waste, where rural areas' waste collection exhibited a fuel consumption of 6-10 L per ton of waste. The estimated fuel consumption was comparable to rural areas fuel consumption; however differences in garbage generation characteristics between the U.S. and Denmark, collection frequency, household setup, non-collection driving activities, and PP_R are responsible for the fuel consumption variability.

Fuel consumption was calculated as L per MT of garbage; however, this analysis cannot be used to compare RCC programs at different PR_R. Fuel consumption should be adjusted to account for the reduction in garbage collection as PR_R increases. As PR_R increases, collected garbage decreases, and the fuel consumed and collection time decease by the garbage fraction in the total waste stream. Garbage collection costs were estimated for RCC programs at two different fuel prices ($1 and $2 per liter of diesel) and are shown in Figure 3.9. The figure also shows the potential savings in garbage collection as PR_R increases from 0% to 100%. An increase in garbage collection services from one to two days is associated with increased fuel, labor, and maintenance cost resulting in 50% increase in collection costs. Doubling fuel price results in a 35% increase in garbage collection costs. Potential savings in garbage collection are considerably higher for programs implementing SS recycling programs for all PR_R because SS programs are more efficient in diverting recyclables from the waste stream, generating less garbage.

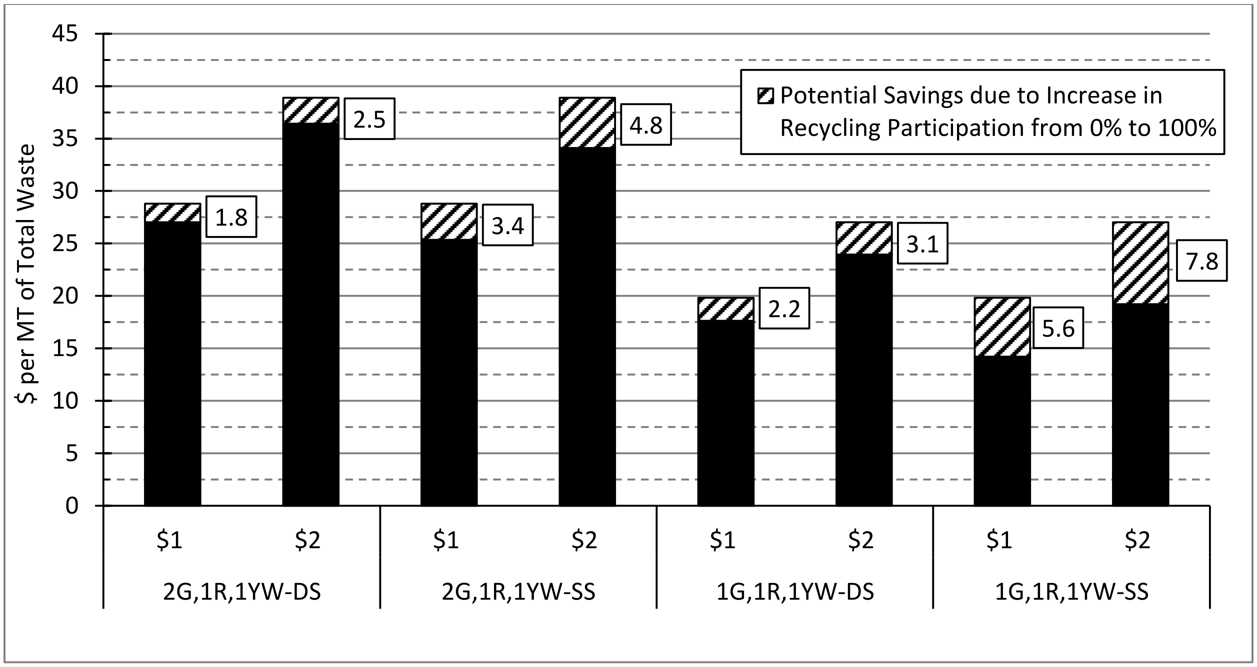

Figure 3.9: Garbage line collection cost. (The collection cost of garbage was estimated for programs with one or two days of garbage collection at two different fuel prices: $1 per liter and $2 per liter. Potential garbage collection cost savings show the reduction in collection cost as recycling participation rate increases from 0% to 100%).

For recyclables collection, the number of households served per trip decreases as PR_R increases. Although the average recyclables collection speed decreases, the fuel consumed (liters diesel per MT of garbage) and collection time (hours per MT of garbage) decreases as PR_R increases. The fuel consumption associated with SS recyclables collection decreases from 48.2 to 19.8 L per MT of recyclables, while total collection time decreases from 3.8 to 1.3 hours per MT of recyclables as PR_R increases from 10 to 100%. For DS recyclables collection system, the fuel consumption decreases from 155 to 45 liters per MT of recyclables, while the total collection time decreases from 10.8 to 3 hours per MT of recyclables. The fuel consumption associated with DS was also reported to be considerably higher than SS collection (42 liters of diesel per MT of recyclables for DS compared to 29 for SS) (Curtis and Dumas, 2000). Moreover, the fuel consumption reported by Curtis and Dumas (2000) was consistent with this study estimates of fuel consumption at higher PR_R values; however a significant increase in fuel consumption was observed at lower PR_R in this study.

Fuel consumption was calculated as L per MT of recyclables; however, this analysis cannot be used to compare RCC programs at different PR_R. Fuel consumption should be adjusted to account for the increases in recyclables collection as PR_R increases. As PR_R increases, the collected recyclables increases, and the consumed fuel and collection time increases by the fraction of the recyclables in the total waste stream. Figure 3.10 shows the net revenues of recyclables collection for RCC programs at three scenarios ($50, $100 and $150 per ton of recyclables) and two fuel prices ($1 and $2 per liter). Revenues were estimated as a function of PR_R for programs using DS or SS recyclables collection systems. As shown in Figure 3.10, the SS recyclables collection systems outperform DS systems for all scenarios. This is due to the high collection time of the DS system which can lead to fuel, labor, and maintenance costs that cannot be compensated by the sale of the collected recyclables. Additionally, SS systems collect more recyclables per stop than DS systems, generating more revenue. An increase in PR_R for DS at moderate recyclables revenues ($100 per ton) will result in further costs associated with collection time that cannot be compensated by selling recyclables. On the other hand, sales of additional recyclables collected by SS systems can compensate for the additional collection time as PR_R increases, except at the lowest recyclables value ($50 per ton) and highest fuel price ($2 per liter).

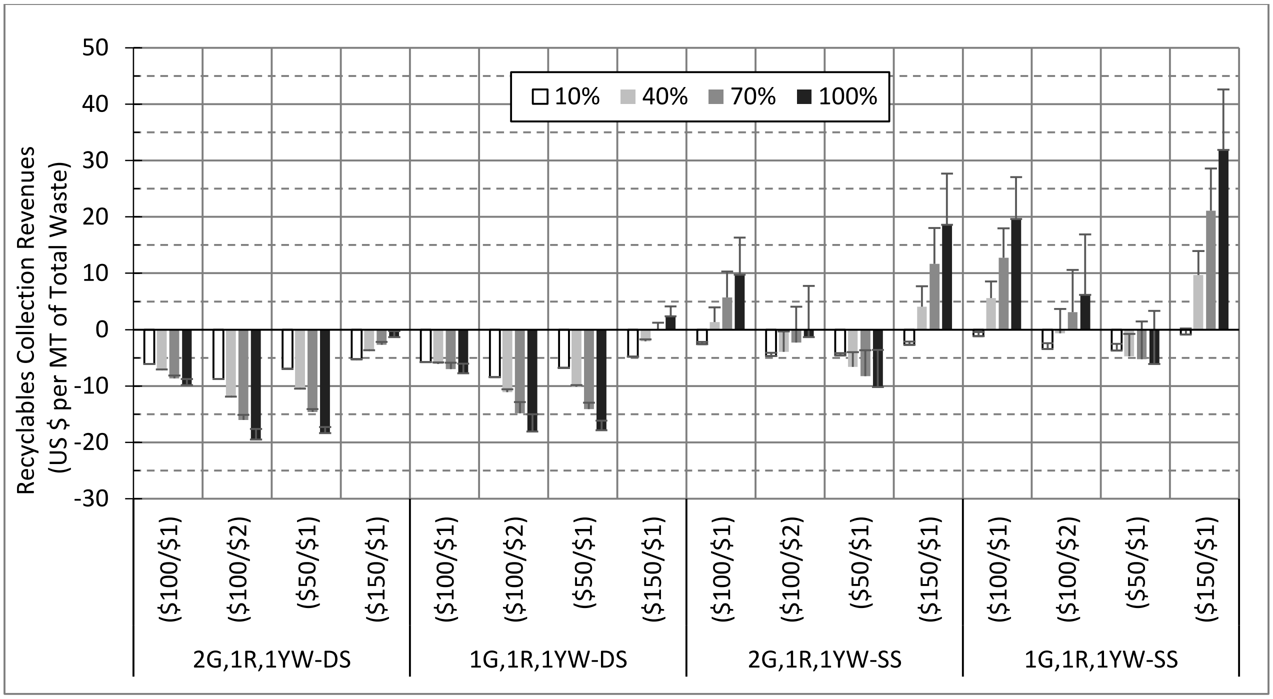

Figure 3.10: Recyclables line collection revenues. Revenues of recyclables collection were estimated for RCC programs at three recyclables net revenues scenarios ($50, $100 and $150 per MT of recyclables) and two fuel prices ($1 and $2 per liter). Whiskers denote potential increase in revenues as a result of recyclables compaction during collection.)

3.2.5. Sensitivity Analysis of Model Parameters and Model Limitations

An analysis was conducted to determine the sensitivity of the results to changing model variables, including the distance between households (D_{HH}), travel speed between households (S_{HH}), and collection time per stop (T_1) (Figure 3.11). The collection time per stop has the greatest effect on collection emissions. For example, a two-fold increase in the collection time increases the collection emissions by 40%. Collection time per stop was based on literature values; however, it can vary based on the number of bins to be collected, collection container, and the collection system technology, e.g., manual, semi, or fully-automated collection.

Travel speed between households was assumed to be independent of the distance between households, which is not necessary true in practice. An increase in the distance between households is usually accompanied by an increase in travel speed. The sensitivity analysis indicated that the effect of collection distance and travel speed on collection emissions are opposite and minimal.

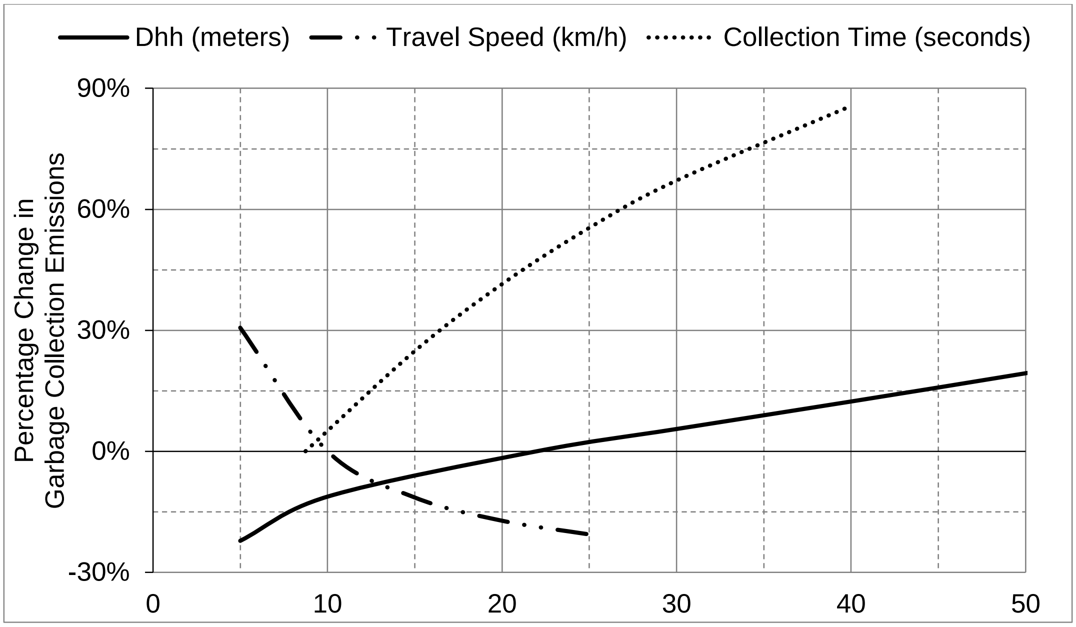

Figure 3.11: Sensitivity analysis of model variables. (Percentage of change in collection emissions due to changing the distance between household (22 to 40m), collection time per stop (9 to 40 seconds) and travel speed between households (5 to 25 Km/h).)

3.4. Conclusions

The study explored the trade-offs between environmental and economic factors of RCC systems in Florida by evaluating the RCC system design of 25 different Floridian communities. An environmental-economic assessment model was developed and used to estimate the greenhouse gas (GHG) emissions and cost of RCC programs. The study results showed that RCC scheduling can significantly impact garbage and recyclables generation rates, recycling efficiency, and consequently determine environmental and economic impact of collection systems.

Overall, the mean total household waste (recyclables and garbage) was 3.11 (±0.56) kg per household per day, while the mean recycling efficiencies were 0.3 (±0.08) and 0.13 (±0.04) for single-stream (SS) and dual-stream (DS) recycling programs, respectively. At the current recycling participating rate (PR_R =70%), the use of SS recyclable collection system diverted 30% compared to 13% of the waste stream by DS. These results indicated that implementing SS collection system can have a positive impact toward achieving Florida's recycling goal of 75% waste diversion. On the other hand, reducing garbage collection frequency had positive environmental and economic effects. The study findings supported the current trends in switching to SS recycling system combined with larger recycling toters, and reduced garbage collection frequency. In comparison with the other European studies (Williams and Cole, 2013), Florida and other U.S. studies (Fitzgerald et al., 2012) showed a significant increase in recyclables generation rate as a result of switching to SS collection. In this study, the same remanufacturing losses per material were applied for SS and DD as specified by WARM; however, the use of SS might result in more contamination and more losses during remanufacturing. This is beyond non-recyclables "waste residue" in the stream and further research is needed. Moreover, this study did not account for emissions associated with overseas shipping of recyclables.

PR_R was found to have a significant impact on the environmental and financial performance of RCC programs. An increase in PR_R reduces garbage collection over a single trip, allowing for serving more households. As a result, emissions associated with the collection of each MT of garbage increases. On the other hand, the fraction of garbage in the total waste decreases, and the emissions associated with garbage collection per MT of total waste decline. For recyclables, the number of households served for recyclables per trip decreases as PR_R increases. Although recyclables collection speed decreases as PR_R increases, it was observed that GHG emissions associated with the collection of each MT of recyclables decreases. Overall, the fraction of recyclables in the total waste increases, and the emissions associated with recyclables collection per MT of total waste increase. Overall, recycling benefits increased substantially at higher recycling participation rate, while collection emissions were insignificant compared to the benefits of recycling. An increase in PR_R will have a positive impact on waste diversion, however more research is needed to address the social aspects of recycling behavior in Florida. Moreover, further research is needed to address the relationship between recycling participation and set-out rates in Florida, and their potential impact on recycling.

The fuel mileage of waste collection vehicles increased from 0.2 and 2.6 Km per liter of diesel consumed as the average collection speed increased from 2 to 60 Km per hour. SS collection offers faster collection time per stop than DS collection, reducing collection emissions and cost. Collection time per stop showed a significant impact on collection emissions and cost; therefore, implementing collection methods that minimize collection time per stop can significantly reduce the collection cost and emissions. Possible examples of other approaches are the automation of the collection system, compliance with bin requirement, and grouping waste containers on shared property lines which cut down the number of stops per route by half.

CHAPTER 4: THE WATER FOOTPRINT OF COMMON MUNICIPAL SOLID WASTE MANAGEMENT PRACTICES

4.1. Introduction

In 2012, approximately 251 million tons of municipal solid waste (MSW) were generated in the U.S.; of which about 135 million tons (53.8% of total generation) were discarded in landfills (U.S. EPA, 2014a). Landfilling is the leading waste management practice in the U.S., followed by recycling and recovering (34.5 %), and combustion with energy recovery (11.7 %) (U.S. EPA, 2014a). The effectiveness of MSW management practices has been evaluated in the published literature and assessment models based on their greenhouse gas (GHG) emissions; economic costs; and airborne, soil, and waterborne emissions (Weitz et al., 1999; Weitz et al., 2002; Consonni et al., 2005; Winkler and Bilitewski, 2007; Buttol et al., 2007; Cherubini et al., 2009). However, the direct and indirect impacts of MSW management practices on water resources have been neglected or not fully considered. Today, the world is challenged by a water crisis threatening global peace, health, and economic development (Bigas, 2012). Many parts of the world struggle with limited water resource availability to sustain growing populations, higher consumption rates, pollutant loadings, and demands of industries, energy sectors, and businesses. The recent U.S. droughts, which affected more than 50% of the U.S. (Fuchs, 2012), have drawn attention to the increasing scarcity of water and the need for sustainable water management strategies.

In the past, the waterborne emissions of some MSW management practices have been evaluated (Winkler and Bilitewski, 2007). The direct consumptive uses of water resources have also been calculated for a few MSW management practices, e.g. recycling and combustion (Arena et al., 2003a; Arena et al., 2003b). However, these calculations do not fully reflect the total (life-cycle) effect of MSW management practices on water resources. As water becomes more of a valued commodity in the world (due to scarcity or impaired quality), consideration of MSW management practices with respect to sustainable appropriation of water resources is essential to promote sustainable development. MSW management practices have direct and indirect impacts on local and global water resources. The water consumption at MSW management facilities, e.g. cooling water, has a direct impact on water resources, while virtual water uses (water used or quality effects along the life-cycle of the process) are considered an indirect impact (Hoekstra and Hung, 2002). Some examples of indirect impacts include water demand and quality effects associated with energy production, raw materials acquisition, capital goods manufacturing, air pollution control, ash conditioning, and leachate treatment and associated releases to the environment.

During the last decade, the WFP methodology (Hoekstra, 2003; Chapagain and Hoekstra, 2004; Hoekstra et al., 2009) has been developed to capture and quantify both direct and indirect effects of processes, products, entities, industries, energy sources, and countries on water resources (Chapagaina et al., 2006; Chapagain and Hoekstra, 2007; Hoekstra and Chapagain, 2007; Dominguez-Faus et al., 2009; Gerbens-Leenesa et al., 2009a; Gerbens-Leenesa et al., 2009b; Hadian and Madani, 2014). Some major industries, entities, and corporations are using the WFP concept as a tool for sustainable appropriation of freshwater resources (Pahlow and Mekonnen, 2012). The WFP provides a reliable criterion for evaluating water use efficiency (Hadian and Madani, 2014) by measuring both the direct and indirect use of fresh water over the entire process life-cycle (Hoekstra et al., 2009).

In this paper, a comprehensive WFP calculation methodology for evaluating the effects of MSW management practices on water resources is described. The local and global WFPs of the three most commonly used MSW management practices (landfilling, combustion with energy recovery, and recycling) were then determined and compared. The results are intended to provide good information about the effects of these practice on water resources. Additionally, the estimation and comparison of the WFP components can help justify certain MSW management practices for particular waste materials. Finally, the calculated WFPs of MSW management practices were compared to other environmental burdens.

4.2. Methodology

4.2.1. A General WFP Calculation Method for MSW Management Practices

The total WFP (WFP_T) of a given MSW management practice is the summation of its blue, green and grey WFP components. The blue WFP (WFP_{Blue}) accounts for the consumption of surface and groundwater resources, which is either lost by evaporation or incorporated into the final product (Hoekstra et al., 2009). The blue WFP of MSW management practices accounts for direct water consumption at MSW management facilities, or indirect consumption during the treatment of the solid waste residue generated (ash, sludge), and consumption of goods and energy.

The green WFP (WFP_{Green}) refers to consumption of green water, the fraction of precipitation falling on land that does not run off or recharge the groundwater but is stored temporarily on top of or within the soil or vegetation, and is normally lost through evapotranspiration (Hoekstra et al., 2009). This component is particularly relevant to agricultural and forestry products, and would apply to bioconversion of waste, and recycling of paper, cardboard, and wood goods.

As for landfills, precipitation runoff during construction will be collected as leachate preventing it from reaching surrounding water bodies. After postclosure, a portion of precipitation will runoff from the covered landfill to nearby water bodies, while a very small portion of the precipitation will infiltrate thought the cover and will be collected as leachate. In this study, it was assumed that all the collected leachate will be treated without losses and released back to the environment as an effluent. Evapotranspiration occurs at the landfill at any time, it was assumed that evapotranspiration will take place regardless of the construction of the landfill. Therefore for the purpose of this study, it was assumed that the green WFP of landfills is equal to zero.

The grey WFP (WFP_{Grey}) relates to water pollution and is defined as the volume of freshwater that is required to attenuate pollutants sufficiently to meet existing water quality standards (Hoekstra et al., 2009). In general, the grey WFP of solid waste is contributed by three components; (1) direct water emissions (liner leakage, effluent), (2) grey WFP of goods and energy, and (3) grey WFP of treating generated solid waste residue (ash, sludge).

The grey WFP (liters of water/ton of waste) is calculated by dividing the pollutant load (L, in mass/ton of waste) by the difference between the maximum acceptable concentration (C_{max}, in mass/volume) and its background concentration in the receiving water body ($C_{background}$, in mass/volume) as shown in Equation 4.1 (Hoekstra et al., 2009).

$$WFP_{Grey} = \frac{L}{C_{max} - C_{backgorund}} \tag{4.1}$$

In the case of an effluent discharge into a water body, the effluent load can be determined by multiplying the effluent flow (Eff, in units of volume/ton of waste) multiplied by the difference between the effluent concentration, C_{eff}, and the natural concentration. The grey WFP is calculated using Equation 4.2 (Hoekstra et al., 2009).

$$WFP_{Grey} = \frac{Eff * (C_{eff} - C_{backgound})}{C_{max} - C_{background}} \tag{4.2}$$

To evaluate the grey WFP, contaminants emitted from MSW management facilities and regulated by U.S. Environmental Protection Agency (EPA) were considered in this study (Table

4.1). As water standards and background concentrations vary among states; ambient water standards suggested by the U.S. EPA were used. Background concentrations also vary across the U.S.; therefore, assumed background concentrations were based on literature norms (Table 4.1).

Table 4.1: Suggested Contaminants Maximum and Background Concentrations.

Contaminants	C_{max}	Units	Source	$C_{background}$	Units	Source
BOD	20	mg/L	U.S. EPA (2009)	<2	mg/L	Chapman (1996)
TSS	20	mg/L	U.S. EPA (2009)	10	mg/L	TSS is one of the most variable characteristics of water quality (Chapman, 1996). For this study, $C_{background}$ was assumed 10 mg/L
NH_3	2.8	mg/L	U.S. EPA (2009)	0.2	mg/L	Chapman (1996)
Arsenic	10	ug/L	U.S. EPA (2009)	2 – 3	µg/L	Hall (2002)
Cadmium	8.8	ug/L	U.S. EPA (2009)	< 1	µg/L	Hall (2002)
Chromium	100	ug/L	U.S. EPA (2009)	2 to 3	µg/L	Hall (2002)
Lead	8.8	ug/L	U.S. EPA (2009)	<5	µg/L	Hall (2002)
Mercury	2	µg/L	U.S. EPA (2009)	< 0.1	µg/L	Hall (2002)
Selenium	35	ug/L	U.S. EPA (2009)	< 1	µg/L	Hall (2002)
Silver	0.07	ug/L	U.S. EPA (2009)	< 1	µg/L	Hall (2002)
Zinc	86	ug/L	U.S. EPA (2009)	<3 to 10.5	µg/L	Hall (2002)
Copper	3.7	ug/L	U.S. EPA (2009)	< 7.7	µg/L	Hall (2002)
Iron	300	ug/L	U.S. EPA (2009)	4 to 16	µg/L	Hall (2002)

Some MSW management practices, e.g. landfilling and combustion, have concomitant value by generating electricity, heat, or gas. Landfill gas (LFG) can be recovered from landfills and used to generate electricity or converted into vehicular fuel, while thermal energy can be recovered from waste combustion and used for heating or to generate electricity. Therefore, their WFP_T would be offset to some extent by the WFP of the generated energy (WFP_{offset}). In the case of recycling, manufacturing using recycled material offsets the WFP of virgin material-based manufacturing to an extent. Thus, the net WFP of any MSW management practice is the summation of the blue, green and grey WPFs reduced by the offset WFP (WFP_{offset}) as shown in Equations 4.3 and 4.4.

$$WFP_N = (WFP_{blue} + WFP_{green} + WFP_{grey}) - WFP_{offset} \quad (4.3)$$

$$WFP_N = \quad WFP_T - WFP_{offset} \quad (4.4)$$

In order to estimate the life-cycle WFP of MSW management practices, the cradle-to-gate WFP of input material and energy sources was obtained from multiple literature sources and studies. Unfortunately, it was observed that the WFP term is often misused in the literature. Previously, Gleick (2003) pointed out the confusion in the water resource literature and, in some cases, the interchangeable use of the terms water use, need, withdrawal, demand, consumption, and non-consumptive use. Hadian and Madani (2013) discussed how the lack of consistent and unique framework for evaluating the impacts of different production processes on water resources have resulted in significantly different and sometimes misleading evaluations of impacts on water resources. Likewise, the WFP methodology of Hoekstra et al. (2009) has not been carefully followed in the literature and the WFP term has been used inappropriately in some cases. Thus, great care should be taken before using or comparing published WFPs.

4.2.2. Waste Landfilling, Combustion and Recycling WFPs

As in any life-cycle assessment, WFP calculations must be framed by function, time, and functional units. The default function of the MSW management systems is the disposal of MSW; thus, any material or energy recovered would offset the process WFP. The functional unit used was liters of water per metric ton (L per MT) of waste processed. The time window used for WFP estimation was 100 years. A sensitivity analysis of the WFP of MSW management practices to site-specific variability was conducted. The WFP is highly related to precipitation, temperature, humidity, wind, soil type, and many other site-specific factors. In this study, only the effects of temperature and precipitation were considered as their effects are expected to be the most significant. Five cities across the U.S. were considered with a climate ranging from wet to dry; Miami, FL; Albany, NY; Austin, TX; San Diego, CA; and El Paso, TX.

4.2.2.1. Waste Landfilling WFP

The WFP of waste landfilling consists of the WFP of landfill construction and operational goods; energy; leachate transportation, treatment and associated releases; fugitive leachate (leakage); and energy recovery offsets (if any). Three types of landfill were considered in this study, namely traditional landfill, bioreactor landfill, and ash landfill. A traditional landfill (also called a dry landfill) is designed to minimize moisture infiltration in an attempt to keep the waste isolated, a bioreactor landfill is designed to enhance waste degradation through increasing moisture content, and ash landfills accept only waste residue from waste combustion facilities (U.S. EPA, 2006).

4.2.2.1.1. Landfill Construction and Operational WFP

For each landfilling method, the quantity of each material consumed per unit waste processed (kg per MT of waste) was used to estimate the construction and operational WFP as shown in Equation 4.5. The WFP (L per kg of material) of each material represents the life-cycle WFP of manufacturing and supplying of each commodity used for construction and operation.

$$
\begin{aligned}
WFP_{C\&O} &= \sum_{i=1}^{n} M_i \times (WFP_{Blue} + WFP_{Green} + WFP_{Grey})_{M_i} \qquad (4.5) \\
&= \sum_{i=1}^{n} M_i \times (WFP_T)_{M_i}
\end{aligned}
$$

where:

$WFP_{C\&O}$ = Construction and operational total WFP (L per MT of waste)

i = Number of materials used

M_i = Quantity of material i consumed per unit waste processed (kg per MT of waste)

$(WFP_T)_{Mi}$ = Life-cycle WFP of material i (L per kg of material)

Landfill cell construction and operational unit processes were adopted from Ménard et al. (2004) for traditional and bioreactor landfills. Ménard et al. (2004) assumed the same shape, depth, and height of landfill cells for traditional and bioreactor landfills. The density of landfilled waste was assumed to be 800 and 1000 kg/m^3 for a traditional and a bioreactor landfill, respectively. The faster degradation of waste in a bioreactor provides more waste capacity per available airspace, and therefore smaller cells and less construction and operational material is used than traditional landfilling (Ménard et al., 2004) (Table 4.3). The waste density of waste in an ash landfill is approximately 2000 kg/m^3 (U.S. EPA, 2011b). Thus, the quantity of material used per unit ash processed should be less than a traditional or bioreactor landfill. In this study, the quantity of each material used per unit ash placed in a landfill was assumed to be 0.4 of the quantity of the same material used in traditional landfill (because the waste density in ash landfills is 2000kg/m^3 compared to 800 for a traditional landfill). For HDPE pipes, half of the quantity was assumed as ash landfills are required to collect leachate but not landfill gas. The unit fuel usage during construction and post-closure was adopted from the Municipal Solid Waste Decision Support Tool (MSW-DST), a life-cycle assessment tool developed by the U.S. EPA (U.S. EPA, 2011b).

For a bioreactor, water is added to increase the moisture content to improve waste degradation. Reinhart and Townsend (1997) found that the optimal operation of a bioreactor occurs at a moisture contents between 40% and 70%, by weight. MSW moisture content depends mainly on waste composition, season of the year, and weather conditions (Alexander, 2003). In the U.S., MSW has a moisture content of 15 to 40 percent, with a typical value of 25% (Tchobanoglous et al., 1993). By regulation, the water content of waste in a bioreactor landfill should be 40% at closure, therefore it is estimated that 150 L of water are needed per MT of waste to bring the moisture content from 25% to 40%. Leachate recirculation is usually used to reach the desired moisture content, however it is rarely sufficient in quantity. The total leachate collected during operation was estimated using leachate generation rates from Section 4.2.2.1.2. The water needed to reach 40% for each U.S. region was calculated in Table 4.2 using these data.

Table 4.2: Water Added to a Bioreactor.

City	Precipitation (mm)	Annual Daily Temp Range (°C)	Total Leachate Collected during operation (L per MT)	Water Deficit† (L per MT)
Miami	1452	23 to 32	82	68
Albany	915	-1 to 28	51	99
Austin	799	17 to 36	45	105
San Diego	236	18 to 25	13	137
El Paso	196	14 to 36	11	139

†moisture needed (150 L per MT) – total leachate collected during operation

The WFP for most construction materials were obtained from the literature. A breakdown of the life-cycle components of each landfilling method is provided in Table 4.3. The WFP of some construction materials was not found in the literature. They were believed to be negligible compared to the total material used and energy inputs, however the WFP may be underestimated.

Table 4.3: Tradition, Bioreactor, and Ash Landfills Construction and Operational Life-cycle Components (kg per MT of waste processed).

		Materials	Traditional	Bioreactor	Ash	Material WFP_T (L per kg)	Reference
			kg per MT	kg per MT	kg per MT		
Operation and Construction	Bottom Layer	Geosynthetic Clay Liner	0.43	0.34	0.21	Unavailable	
		Geomembrane	0.43	0.34	0.21	Unavailable	
		Geonet	0.59	0.50	0.29	Unavailable	
		Bentonite	0.03	0.00	0.01	Unavailable	
		Geotextile	0.05	0.04	0.02	Unavailable	
	Leachate & LFG Collection	Gravel	105	101	53	0.11	Climate Change Research (2010)
		HDPE Pipes	0.44	0.10	0.22	237	Van der Leeden et al. (1990)
	Leachate Recirculation	Steel Tank	n/a	0.06	0	70.9	Van der Leeden et al. (1990)
		Aluminum Dome	n/a	0.00	0	57.4	Li (2010)
		Reinforced Concrete Base	n/a	0.50	0	0.8	Assumed similar to precast concrete (Building Research Establishment, 2007)
	Final Cover	Sand	107	90	54	0.01 to 0.11	Climate Change Research (2010)
		Organic Soil	22	19	11	Unavailable	
	Onsite Diesel Use	Diesel (non-road Equipment)	1.0	1.0	0.7	38	Crude oil world average WFP 1.06 m^3/GJ†
	Bioreactor	Water Added	n/a	68-139	n/a	1	
Postclosure		Sand for Final cover replaced	10	10	5	0.11	Climate Change Research (2010)
		Total Postclosure Fuel Used (Liters)	6.2E-07	6.2E-07	6.2E-07	38	Crude oil world average WFP 1.06 m^3/GJ †

n/a: not applicable

† Gerbens-Leenes, 2008; Gerbens-Leenes et al., 2009a

4.2.2.1.2. Leachate Generation Rate and Quality

Waste landfilling pollutes infiltrating rainwater that is collected as leachate, preventing it from reaching local surface and groundwater reservoirs. The bulk of leachate is collected and treated, following which it is released back to the local environment. Therefore, the green WFP of waste landfilling associated with cover storage was assumed zero. To evaluate the average quantity of leachate released during the lifetime of one ton of waste, it was assumed that a landfill has a useful life of 20 years and a ton of waste is placed in the middle of the useful life of the landfill (after 10 years) (U.S. EPA, 2011b). Thus in 100 years, leachate is collected and treated on average for 40 years (10 years after placement until closure and 30 years during postclosure), and is released to the environment without treatment afterward. During this period, leachate generation rate can be divided into three phases; phase I (0 to 1.5 years) is during cell construction with daily cover; phase II (1.5 to 10 years) is when the cell receives intermediate cover; and phase III (10 years to 100 years) follow final cover placement. Leachate generation rate is highly related to precipitation, temperature, humidity, wind, soil type, and many other site-specific factors. The U.S. EPA (2011) estimated leachate generation to be 20%, 6.55%, and 0.04% of the precipitation during Phase I, II and III, respectively (U.S. EPA, 2011b). Accordingly, the annual leachate generation rate (L per MT per year, Q_t) was estimated during each phase using Equation 4.6. It was assumed that the depth of the cell is 15m. The calculated Q_t values during each phase for each region are listed in Table 4.4.

$$Q_t = \frac{F_t x \left[\frac{P}{1000}\right] x A}{A x D x \rho} x \frac{1000kg}{1\ MT} x \frac{1000L}{m^3} = \frac{F_t x P x 10^6}{D x \rho} \qquad (4.6)$$

Where:

P = Annual precipitation (mm/year)

F_t = Percent of precipitation collected as leachate during each phase (%)

A = Cross section area of the landfill (m^2)

ρ = Density of the landfilled waste (kg/m^3)

D = Waste depth (m)

Table 4.4: Average Annual Leachate Generation Rate (Q_t, L per MT per year).

City	Precipitation (mm)	Traditional			Bioreactor			Ash		
Phase		I	II	III	I	II	III	I	II	III
Miami	1452	23.9	7.8	0.048	19.1	6.3	0.038	9.5	3.1	0.019
Albany	915	15.0	4.9	0.030	12.0	3.9	0.024	6.0	2.0	0.012
Austin	799	13.1	4.3	0.026	10.5	3.4	0.021	5.3	1.7	0.011
San Diego	236	3.9	1.3	0.008	3.1	1.0	0.006	1.6	0.5	0.003
El Paso	196	3.2	1.1	0.006	2.6	0.8	0.005	1.3	0.4	0.003

BOD, TSS, ammonia, and heavy metal concentrations were used to estimate the grey WFP of landfills. In ash landfills, the BOD of leachate was assumed to be zero due to the lack of biodegradable organics (U.S. EPA, 2011b). In a traditional landfill, the BOD was assumed to be about 10,000 mg/L during phase I, and then drop linearly to 1,000 mg/L by the end of phase II. After phase II, BOD drops linearly to 10 mg/L by year 50 and stabilizes afterward. The BOD of a bioreactor landfill leachate decreases at a faster rate. After year 1, the BOD concentration decreases linearly from 10,000 to 1,000 mg/L in two years. Afterward the BOD concentration decreases linearly to reach 10 mg/L in seven years, where it remains afterward (U.S. EPA, 2011b). The concentrations of TSS, NH_3, and heavy metals in leachate from traditional, bioreactor and ash landfills were assumed to be constant over time (U.S. EPA, 2011b). The low and high median concentrations of TSS, NH_3, and heavy metals in landfill leachate for traditional, bioreactor, and ash Landfills were adopted from the Environmental Research and Education Foundation (EREF) (1997) and the U.S. EPA (1990) reports (Data provided in Table A.4 in the Appendix). The upper median concentration was used to calculate the grey WPF associated with each contaminant.

Leachate leaking from the bottom liner was assumed to have different leachate characteristics during each of the described phases above. The grey WFP associated with leachate leaked can be calculated using Equation 4.7 and Table 4.1 and Table **4.4**, assuming that 0.2% of leachate leaks through the liner (U.S. EPA, 2011b). The grey WFP of leachate leaked is selected based on the contaminant with the largest grey WFP.

$$WFP_{G\_Leakage} = \sum_{t=0}^{100} \frac{0.2\% * Q_t * (C_L - C_{backgroundl})}{C_{max} - C_{Background}} \quad (4.7)$$

where:

$WFP_{G\_Leakage}$ = Grey WFP of contaminant leaked (L per MT of waste)

t = Number of years considered (100 years)

C_L = Contaminant concentration in leachate during each year (mg/L)

4.2.2.1.3. Landfill Gas (LFG) Recovery

LFG can be recovered from landfills and used to generate electricity, among other uses. Thus, WFP_T of waste landfilling would be offset to some extent by the WFP of the generated energy. To estimate LFG generation, the U.S. EPA LandGEM Landfill Gas Emissions Model (version 3.02) was used. LandGEM requires the user to specify two inputs: the potential methane generation capacity (L_0, m^3 per MT) and methane generation rate constant (k, $year^{-1}$) (Alexander et al., 2005).

In Florida, Amini et al. (2012) estimated L_0 to vary from 56 to 77 m^3 per MT of waste, while k value estimates ranged from 0.04 to 0.13 $year^{-1}$ for traditional landfills and was 0.10 $year^{-1}$ for a bioreactor. Barlaz et al. (2010) also observed faster methane generation rate in a bioreactor landfill (k = 0.08-0.21 $year^{-1}$). For an optimal moisture content, it is expected that k values vary among locations with different climates. The range of high temperatures observed at each city is listed in Table 4.5. For the purpose of this study, it was assumed that an upper end k value (k = 0.1

year^{-1} within the range observed by Amini et al. (2012) in Florida) will be observed in a warm city like Miami, while a lower end k value (k = 0.08 year^{-1}) will be observed for a bioreactor landfill in a cold city such as Albany. An average k value of 0.09 was used for San Diego, El Paso and Austin.

For a traditional landfill, the AP-42 recommends a default k value of 0.02 and 0.04 year^{-1} for areas receiving below 635 mm and greater than 635 mm of annual rainfall, respectively. In this study, k value was assumed to be equal to 0.04 for cities receiving more the 635 mm of rain, and 0.02 for other regions (U.S. EPA, 2008b). L_0 was assumed to be 70 m^3 per MT (within the range observed by Amini et al. (2012) in Florida landfills) for all landfills. A summary of assumed k values is shown in Table 4.5. The cumulative LFG production in 50 years (20 years of landfill lifetime and 30 years of postclosure period) was estimated using the U.S. EPA LandGEM Landfill Gas Emissions Model (version 3.02) and listed in Table 4.5. The lifetime collection efficiency of LFG was assumed to be 75% (reported collection efficiencies typically range from 50 to 95%, with a default efficiency of 75% (U.S. EPA, 1997)) and the efficiency of electricity generation was assumed to be 33% (U.S. EPA, 2011b). It was assumed that the LFG is 50% methane, and the energy content of one m^3 of methane gas is equal to 10 kWh.

According to the U.S. EPA (2012d), large landfills (traditional or bioreactor landfills) are required to collect LFG to be beneficially used or flared. The collection of LFG resulted in condensate generation, which is collected with leachate and either sent to a municipal wastewater treatment facility or recirculated back into the landfill. The amount of condensate production (Q_{Cond}) from three sites with LFG recovery systems ranged from 44 to 162 liters per l000 cubic meters of unprocessed LFG and was not correlated with climate (Briggs, 1988). Table 4.5 lists the potential quantity of condensate collected, assuming a 75% collection efficiency for LFG. For a traditional landfill, it was assumed that all condensate is sent to a wastewater treatment facility, while for a bioreactor it was recirculated.

Table 4.5: Potential LFG Recovery from Traditional and Bioreactor Landfills.

City		Traditional				Bioreactor			
	Annual Daily Temp Range (°C)	k ($year^{-1}$)	LFG (m^3 per MT)	Condensate (Q_{Cond}, liters per MT)	Potential Energy (kWh per MT)	k ($year^{-1}$)	LFG (m^3 per MT)	Condensate (liters per MT)	Potential Energy (kWh per MT)
Miami	23 to 32	0.04	112	13.6	139	0.1	138	16.8	170
Albany	-1 to 28	0.04	112	13.6	139	0.08	134	16.4	165
Austin	17 to 36	0.04	112	13.6	139	0.09	136	16.5	168
San Diego	18 to 25	0.02	77	9.4	95	0.09	136	16.5	168
El Paso	14 to 36	0.02	77	9.4	95	0.09	136	16.5	168

4.2.2.1.4. Leachate Transportation and Treatment

In this study, it was assumed that leachate and condensate will be collected from traditional landfills and transported to a local wastewater treatment facility until the end of the postclosure period, for a total of 40 years. For a bioreactor, it was assumed that all leachate is recirculated back into the landfill, until the end of the postclousre period for a total of 40 years. The WFP of leachate transportation can be calculated using Equation 4.8. It was estimated that the fuel consumption per kg of leachate transported (FC_L) is equal to 0.89 liters of diesel per MT of leachate transported (U.S. EPA, 2011b).

$$WFP_{T\_Trans} = (Q_{Cond} + \sum_{t=0}^{40} Q_t) * D_L * FC_L * WFP_{T(D)} \quad (4.8)$$

where:

WFP_{T_Trans} = Total WFP of leachate transportation (L per MT of waste)

D_L = Density of leachate (1 MT per m^3)

$WFP_{T(D)}$= Total WFP of diesel fuel (Liters of water per liter of diesel)

t = years

The WFP of leachate treatment consists of three components: the WFP of electricity used to treat leachate; the wastewater effluent's grey WFP, and the sludge WFP back in the landfill. The WFP of sludge treatment was considered to be minimal and was not considered in this study. The electricity consumption associated with leachate treatment is a function of BOD removal efficiency. The default treatment efficiencies of an average POTW (Robinson and Knox, 2003; U.S. EPA, 1998; U.S. EPA, 1992) as complied by the U.S. EPA (2011b) were 97, 80, 98, 21.6, 96, and 85% for BOD, COD, NH3, TSS and heavy metals, respectively. The WFP of electricity can be calculated using equation 4.9.

$$WFP_{T\_Electrcity} = \sum_{t=0}^{40} 0.97 * Q_t * C_{(BOD)_t} * EC * WFP_{T(US\ Electricty)} \qquad (4.9)$$

where:

$WFP_{T\_Electrcity}$ = Total WFP associated with leachate treatment electricity used (L per kwh)

$C_{(BOD)t}$ = Leachate BOD concentration (mg per L) during each year (t)

EC = Electricity Consumption per BOD removed (1 kWh per kg BOD removed (U.S. EPA, 2011b))

$WFP_{T(U.S.\ Electricity)}$ = Average total WFP of US electric grid (9 L per kWh (refer to Table A.5 in the Appendix)

The grey WFP associated with each contaminant in the treated effluent can be calculated as shown in equation 4.10. The grey WFP of the effluent is selected based on the contaminants with the largest grey WFP.

$$WFP_{G\_Effluent} = \sum_{t=0}^{40} \frac{(1-n) * Q_t * (C_L - C_{background})}{C_{max} - C_{backgorund}} \qquad (4.10)$$

where:

$WFP_{G\_Effluent}$ = Grey WFP of POTW effluent (L per MT waste)

4.2.2.2. Waste Combustion WFP

A detailed breakdown of the life-cycle components of a waste combustion facility (construction and operating goods, energy use, and products) is provided in Table 4.6. The components used in the construction of the combustion facility were adopted from Brogaard et al. (2013). The operational goods used were based on the U.S. EPA MSW-DST values and other sources. The construction and operating WFP components of a waste combustion facility were calculated using Equation 4.5. The total WFP of ash landfilling was estimated to be between 82 and 180 L per MT, depending on landfill location using assumption for 4.2.2.1. The WFP of ash landfilling was added to the WFP of waste combustion facilities.

Table 4.6: Life-cycle Components of Waste Combustion Management Facility.

Process Unit Inputs and Outputs	Quantity (kg per MT Waste Processed)	Reference	WFP_T (Liters of water per kg)	Reference
Construction Phase				
Concrete	4-13	Brogaard et al. (2013)	0.8 Liter per Kg	BRE (2007)
Steel	0.37-0.98	Brogaard et al. (2013)	65.2 L per Kg	Van der Leeden et al. (1990)
Glass	0.001-0.008	Brogaard et al. (2013)	Unavailable	
Fiberglass	0.03	Brogaard et al. (2013)	3.58 L per Kg	Niccolucci et al. (2011)
Machinery Steel	0.94-1.59	Brogaard et al. (2013)	65.2 L per Kg	Van der Leeden et al. (1990)
CMS and HVCs	0.08-0.18	Brogaard et al. (2013)	Unavailable	
Other Steel	0.04-0.08	Brogaard et al. (2013)	65.2 L per Kg	Van der Leeden et al. (1990)
Other Concrete	0.1-0.2	Brogaard et al. (2013)	0.8 Liter per Kg	BRE (2007)
Asphalt	0.06-0.11	Brogaard et al. (2013)	Unavailable	
Energy (Wh) (on-site construction)	0.7-1.9	Brogaard et al. (2013)	9 Liter per Kwh	Refer to Table A.5
Operational Phase				
Process Water (Cooling)	158	Arena et al., 2003b	1 Liters per Kilogram	Assumed one liter of water per liter of water consumed, however piping, pumps and energy used might contribute to a bigger WFP.
Lime	7.1	Harrison et al., 2000	Undefined	
Ammonia	1.5	Harrison et al., 2000	7.31 L per Kg	Tata Group (2013)
Carbon	0.4	Harrison et al., 2000	Undefined	
Ash Cooling Water	17.2	Arena et al., 2003b	1 Liters per Kilogram	Assumed one liter of water per liter of water consumed, however piping, pumps and energy used might contribute to a bigger WFP.
Sodium Silicate (Ash Conditioning)	1.5	Arena et al., 2003b	Undefined	
Cement (Ash Conditioning)	13.5	Arena et al., 2003b	3.29 L per Kg	Tata Group (2013)
Products				
Electricity Production (kwh)	550 (Net electricity Production)	Net electricity Production (Harrison et al., 2000)	9 L per Kwh	Refer to Table A.5
Ash	50-122	Harrison et al. (2000), and Arena et al. (2003b)	0.08-0.18 L per kg	This study
Potential Steel and Iron Recovery	5 to 43 (90% recovery assumed)	(90 recovery assumed, Harrison et al., 2000)	65.2 L per Kg	Van der Leeden et al. (1990)

4.2.2.3. Recycled Commodities WFP

Recycling recovers materials that can be remanufactured or substituted for virgin material manufacturing. Therefore, recycling will offset the WFP of virgin material-based manufacturing to some extent. The WFP offsets of recycled commodities were calculated using Equation 4.11. In general, a portion of the recovered material is not suitable for use as a manufacturing input, and is discarded at the material recovery facility or the manufacturing stage (U.S. EPA, 2012a). Four recycled commodities were considered in this study namely aluminum cans, steel cans, HDPE, and office paper. The percent material losses (x) during recycling were based on data compiled by the U.S. EPA and Research Triangle Institute in the US EPAWARM version 13 software, which suggested x to be 7%, 2%, , 14%, and 40% for aluminum cans, steel cans, HDPE, and office paper, respectively.

$$WFP_R = WFP_m - (1 - x) * (WFP_v) \quad (4.11)$$

where:

WFP_R = Net WFP of each recycled material (L per kg recycled)

WFP_m = Total WFP of remanufacturing (L per kg recycled)

WFP_v = WFP of virgin material manufacturing (L per kg of virgin material)

4.3. Results and Discussion

This section presents the calculated WFPs of waste landfilling, combustion and recycling of some commodities. The WFP of waste landfilling was calculated for traditional and bioreactor landfills with and without energy recovery, while the WFP of waste combustion was calculated for a facility recovering energy and ferrous materials (tin cans). In the case of waste combustion, the WFP of ash landfilling was added to the total WFP of the facility. Finally, the WFP of recycling was presented for some recycled commodities.

4.3.1. Waste Landfilling WFP

The total WFP of waste landfilling consists of the WFP of construction and operational goods, leachate transportation, leachate treatment and associated releases, and fugitive leachate (leakage). The blue and grey WFPs of waste landfilling methods are shown in Figure 4.1. The blue WFP of waste landfilling is mainly related to construction and operating capital goods, except for bioreactor landfills where added water is the most significant component.

The grey WFP of waste landfilling technologies consists of the WFP of construction and operation goods, untreated leachate discharge, and liner leakage. The grey WFP of waste landfilling is highly dependent on the waste type, location, and technology. The grey WFP of bioreactor landfills was always less than traditional landfills. For a traditional landfill, the grey WFP is a function of precipitation; grey WFP decreases as precipitation decreases. Generally, the grey WFP of a traditional or bioreactor landfill leakage and treated effluent was attributed to BOD, while for ash landfill it was attributed to arsenic concentration. After the end of the postclosure period, the grey WFP was attributed to ammonia leaching for traditional and bioreactor landfills, and arsenic for an ash landfill.

The blue and grey WFPs of traditional landfills were 70-79 and 360-1,800 L per MT of waste processed, respectively. The use of bioreactor landfills reduced the grey WFP of landfilling by 80%, while it increased the blue WFP of waste landfilling to 200 L per MT in dry areas. Using bioreactor landfill technology reduces the volume of leachate to be treatment, resulting in a much smaller grey WFP for landfilling (Figure 4.1). However, the water added to a bioreactor located in dry climates contributed to a larger blue WFP compared to a traditional landfill in the same climate. Overall, the use of bioreactor landfills reduced the total WFP of waste landfilling by 73% in Miami (wet climate), while the decrease was about 35% in El Paso (dry climate) mainly due to adding more water and the smaller grey WFP of traditional landfills in dry climates. The total WFP of energy used for leachate transportation and treatment was less than 1% of the total WFP of scenario. Finally, the blue WFP of ash landfilling was about 42 L/MT, while the grey WFP ranged between 40 to 138 L/MT.

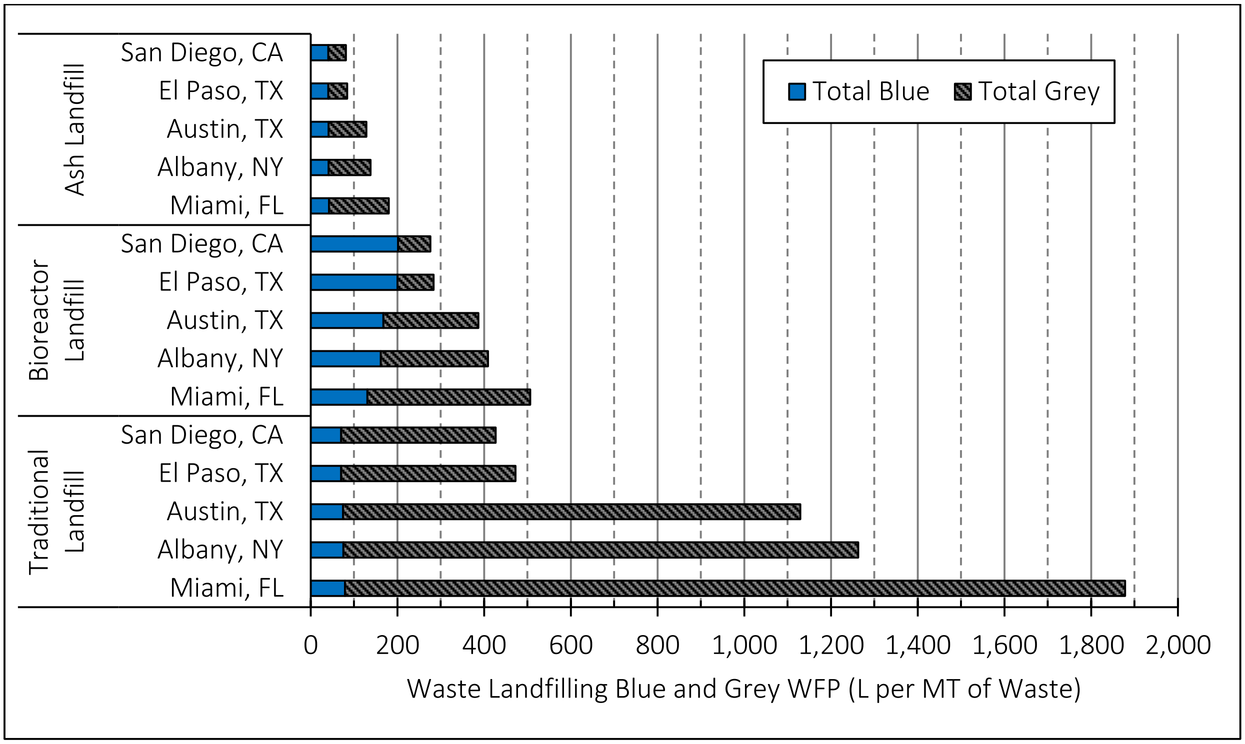

Figure 4.1: Blue and Grey WFPs of Waste Landfilling Methods over 100 Years of Landfill Life.

Traditional and ash landfills do not necessarily impact local blue water resources; however, the majority of the grey WFP impact is local. Figure 4.2 illustrates the grey WFP of components of waste landfilling technologies. Treated effluent is the most significant component of grey WFP, except when leachate is completely recirculated in bioreactor landfills. The grey WFP of discharging untreated leachate after the postclosure period had the second largest contribution to grey WFP.

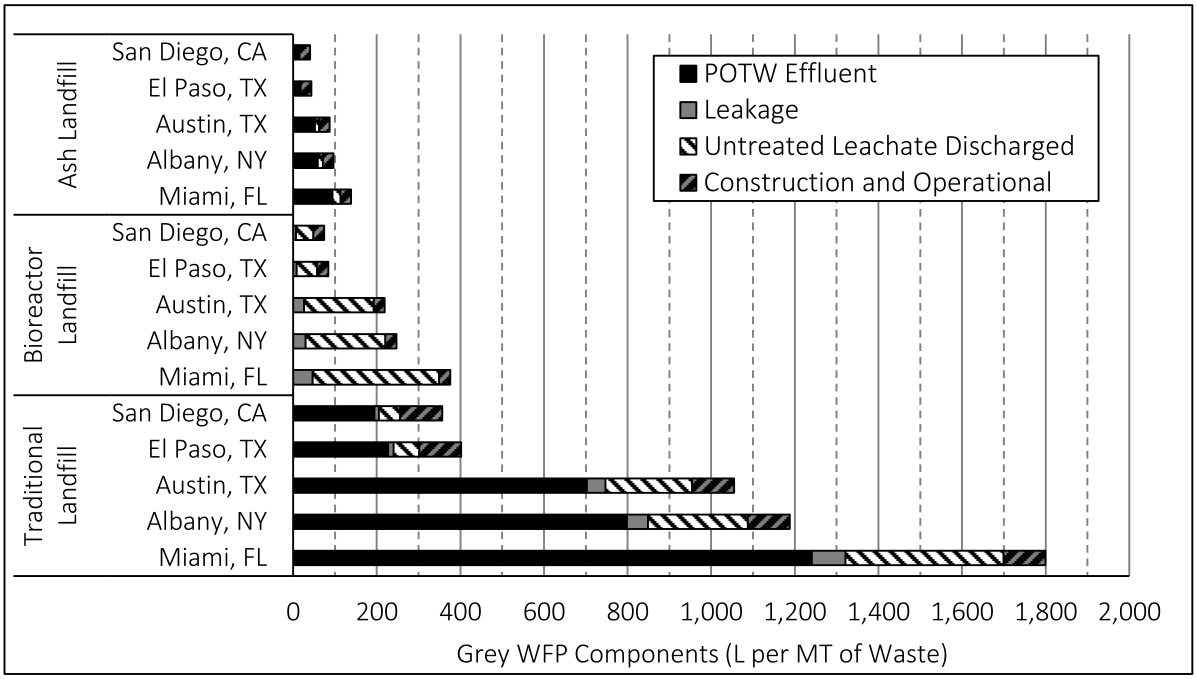

Figure 4.2: Components of the Grey WFP of Waste Landfilling Methods.

4.3.2. Waste Combustion WFP

The blue and grey WFPs of waste combustion facilities are shown in Figure 4.3. The grey WFP was slightly affected by the location of the ash landfilling facility, ranging between 126 and 134 L per MT of waste. The blue WFP of waste combustion was mostly associated with cooling water which accounts for 63% of the blue WFP, followed by cement that is used for ash conditioning (18%), and ash cooling water (7%). On the other hand, construction steel accounted for 91% of the grey WFP, while ash landfilling accounted for 9% of the grey WFP. Blue WFP accounted for 68% of the WFP, grey 32%. Cooling WFP accounted for 45% of the total WFP, while steel accounted for 33%.

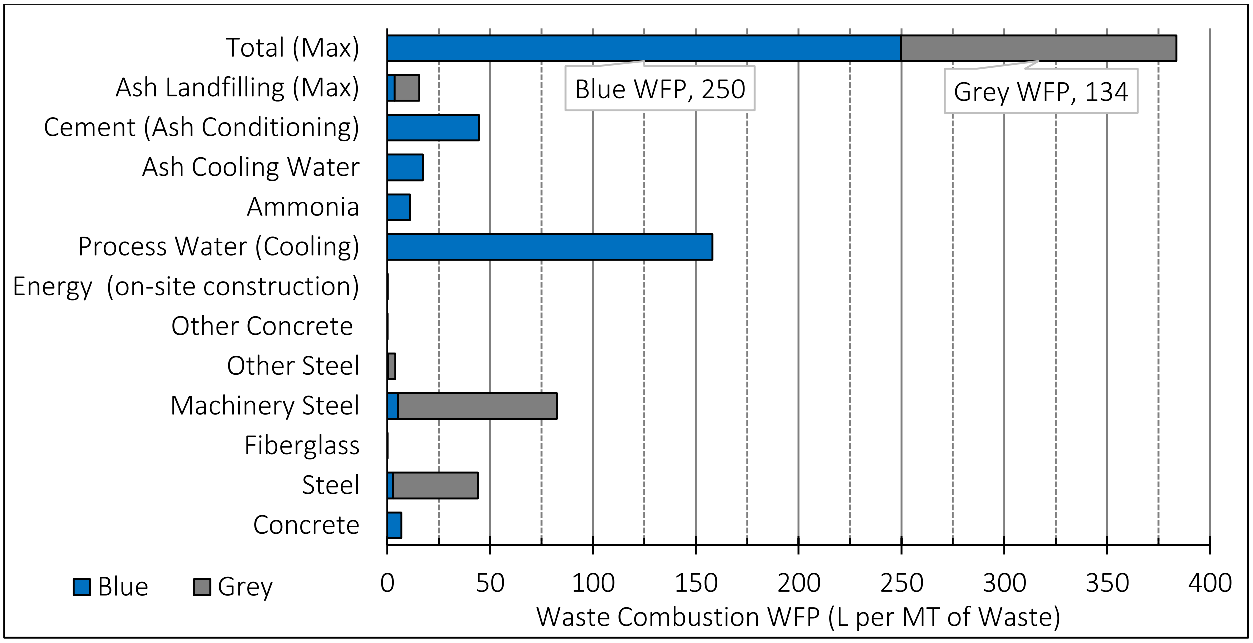

Figure 4.3: WFP of Waste Combustion Plant Construction and Operation.

To generate electricity from waste combustion, water is heated, turning into steam which drives turbines to generate electricity. Afterward, steam is condensed using an open loop and closed loop cooling system (Feeley et al., 2008). An open loop system withdraws water once from a local water body, then the warmed water is discharged back into the environment. The closed loop (recirculating) is either wet cooling or dry cooling. In wet cooling, cooling towers are used to dissipate heat from the water to the atmosphere. In general, dry cooling uses a high flow rate of ambient air that is blown by fans to condense steam. Thus the consumptive and withdrawal water uses are minimal (Feeley et al., 2008). In this analysis, processing water accounted for 43% of the total WFP of waste combustion. The use of a more water efficient cooling system such as dry cooling has the potential to reduce the total WFP of waste combustion, however it might reduce the overall efficiency of electricity generation as well as the environmental benefits of waste combustion, e.g. GHG emissions and WFP offsets. Site-specific calculations are required to determine the WFP and energy WFP offsets.

4.3.3. WFP Offsets

The function of MSW management practices is to dispose and treat MSW, thus the electricity generated and any material recovery will offset the WFP_T. In the US, the WFP of the electric gird was calculated to be 9 L per kWh in 2013 (Table A.5 in the Appendix). The total, offset, and net WFPs for traditional landfills, bioreactor landfills, and waste combustion facilities recovering energy are provided in Figure 4.4. For traditional landfills, the WFP offsets of the recovered energy could not completely compensate for the WFP of waste landfilling in wet climates (Miami, FL; Albany, NY), mainly due to the large grey WFP associated with leachate treatment and releases. On other hand, the WFP offsets of the recovered energy compensate for the construction and operational WFP of a bioreactor landfill in any climate. The net WFP of traditional waste landfilling ranged between -429 and 327 L per MT of waste, while the net WFP of bioreactor landfilling ranged between -1,240 and -1,020 L per MT. The WFP offsets associated with recovering energy from waste combustion facilities were significantly larger than recovering electricity from LFG.

A second potential offset to combustion is recovery of ferrous materials. It is estimated that 90% of ferrous materials are recovered at waste combustion facilities; about 5 to 43 kg of steel per MT of waste can be recovered. The total WFP of steel was estimated to be 65.2 L per kg by Van der Leeden et al. (1990). Thus, the WFP offsets of recovering steel could reach 2,800 L per MT of waste, if steel remanufacturing WFP is neglected.

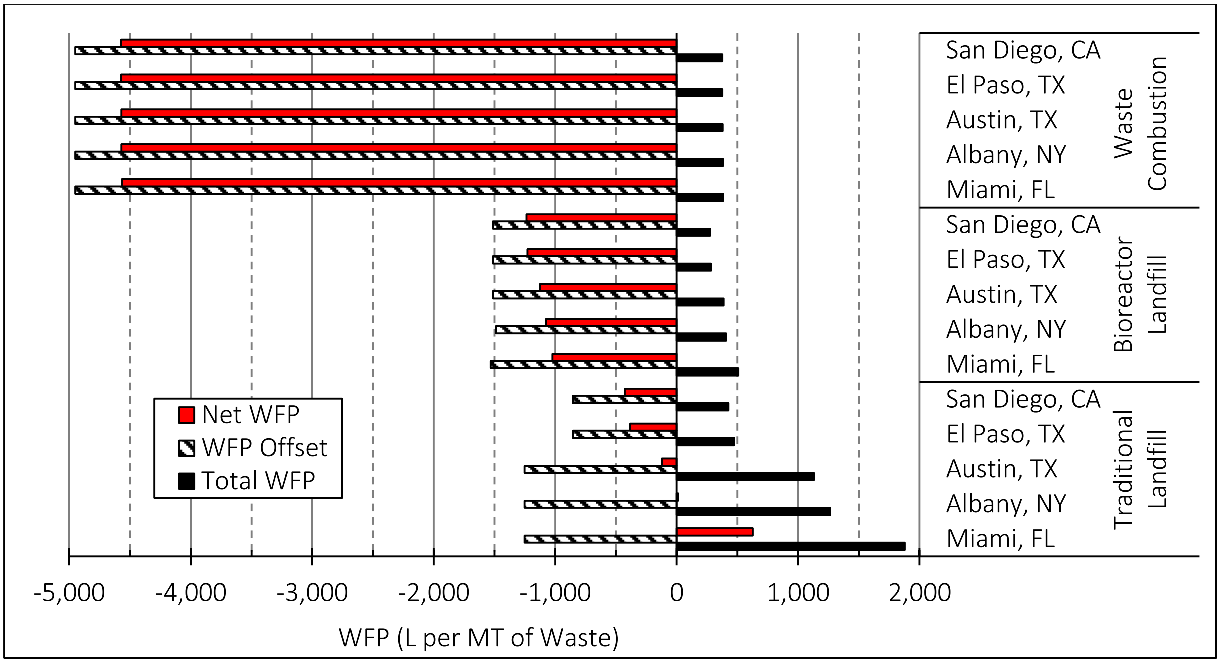

Figure 4.4: The Total, Offset, and Net WFPs of MSW Management Practices with Energy

4.3.4. Comparison of WFP and GHG emissions

In reality, the benefits of energy recovery are going to vary as energy sources, energy mixes, cooling technology, and generation efficiency change across the US and the globe. In an attempt to compare the WFP of the recovered energy to other conventional non-renewable and renewable energy sources, the WFP of the recovered electricity was calculated (L per kWh) for MSW landfills and combustion facilities. The WFP range of electricity recovered from MSW management practices as well as other conventional and non-conventional energy sources adopted from Hadian and Madnai (2013) are presented in Figure 4.5. The GHG emissions (kg CO_{2eq} per kWh) of energy recovered from MSW management practices and energy sources (Kaplan et al., 2009; Fthenakis and kim, 2007; Holm et al., 2012) are also presented in Figure 4.5. The average WFP and GHG emissions of the electric grid are calculated in Table A.5 and Table A.6 in the Appendix. The WFP of the electricity recovered from waste combustion had a lower WFP than the U.S. electric grid, and other conventional energy sources. In comparison with other renewables, the WFP of energy recovered from MSW management practices was also drastically lower than other conventional renewable such as hydropower and biomass which averaged 79 and 250 L per kWh, respectively (Gerbens -Leenes, 2008).

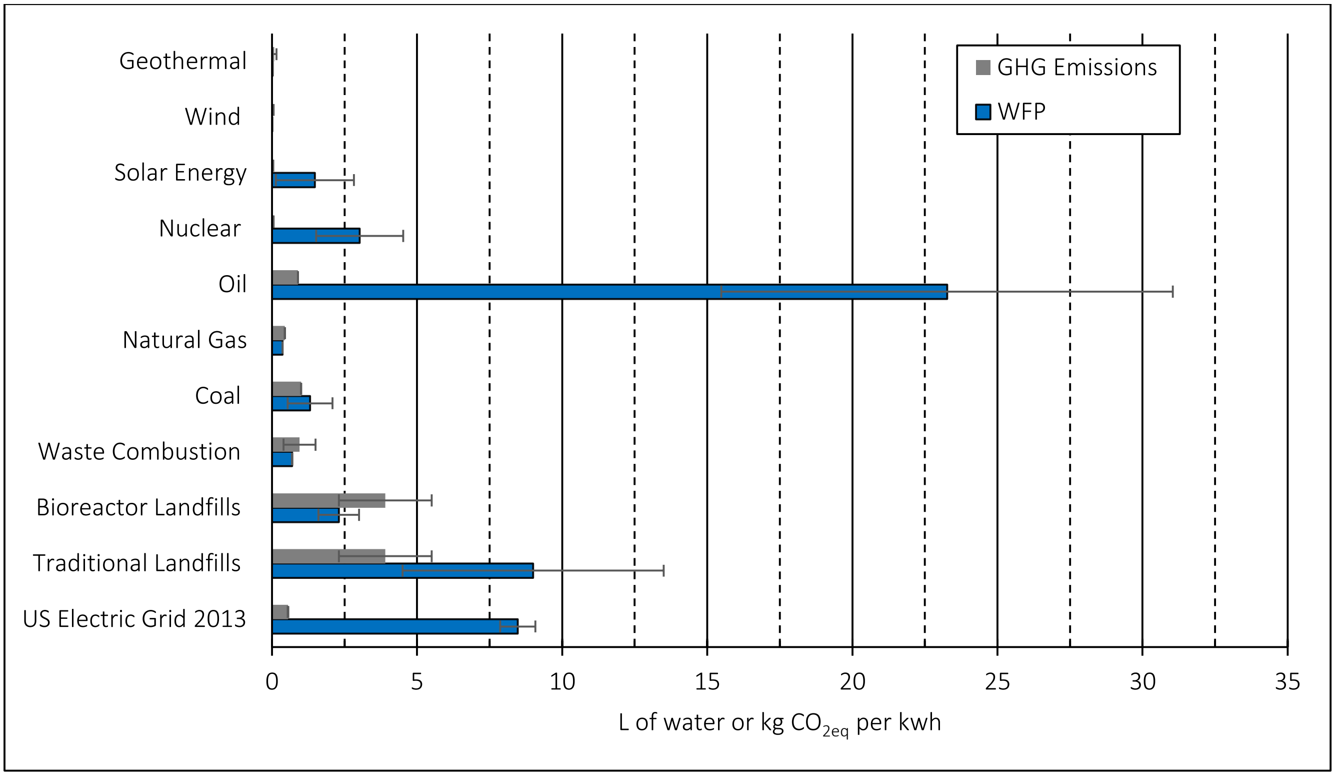

Figure 4.5: The WFP and GHG emissions of electricity recovered from waste compared to other conventional renewable and non-renewable sources (WFP data from (Hadian and Madani, 2013) and GHG emissions data from (Kaplan et al., 2009; Fthenakis and kim, 2007; Holm et al., 2012). Data and references are available in
Table A.7 in the Appendix).

Overall, WFP had similar patterns regarding relative GHG emissions of MSW management practices (i.e., net WFP and net GHG emissions of landfills were greater than waste combustion due to energy benefits); however, landfills had a lower WFP compared to oil and several renewables with respect to electricity generation WFP.

4.3.5. WFP of Recycling

The net WFP and GHG emissions are presented for several commodities as shown in Figure 4.6. The WFP calculations are presented in

Table A.**7** in the Appendix. The U.S. EPA WARM software, version 13, was used to estimate the GHG emissions of landfilling with energy recovery, combustion and recycling of some commodities. The WFP calculations considered the WFP offsets as a result of using recycling material to substitute virgin-based recycling, however the WFP of collecting, processing, and remanufacturing of recyclables were not considered in this study. The recycling of renewable goods, e.g. paper, resulted in the highest WFP offset compared to other recycled materials evaluated. This is mainly due to the high green and grey WFPs of material acquisition and manufacturing of paper goods.

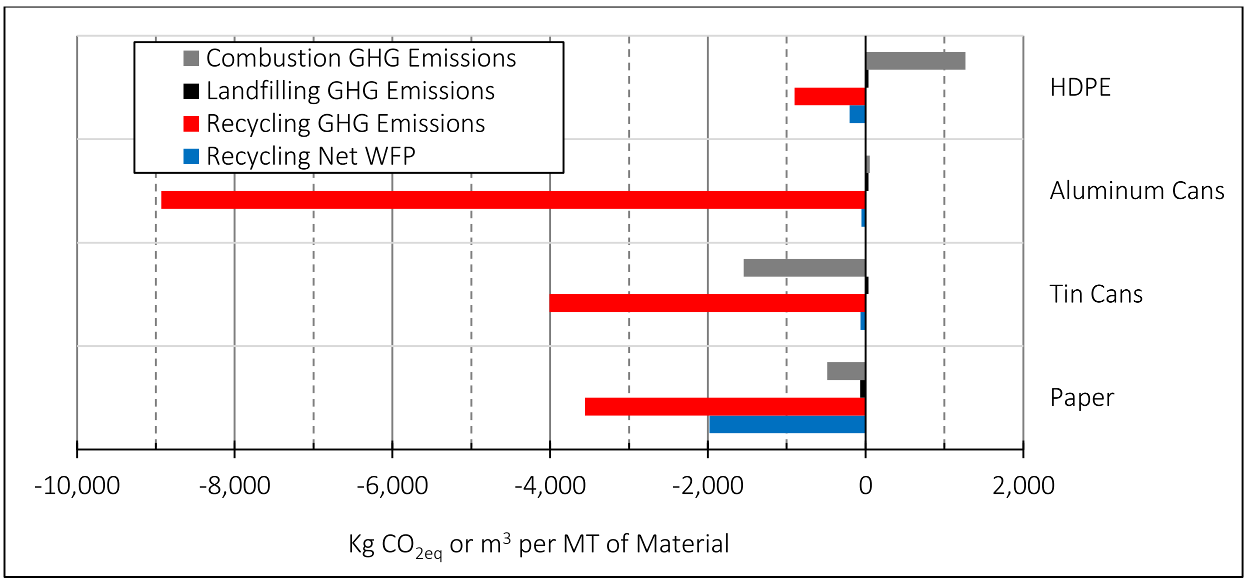

Figure 4.6: Recyclables Net WFP and GHG emissions (WFP data from (UPM-Kymmene, 2011; Van der Leeden et al., 1990; Li, 2010). WFP data are available in Table A.8 in the Appendix).

4.4. Limitations of the Study

Accounting for the WFP criterion in evaluating MSW management practices is important for sustainable appropriation of water resources. This study aimed to illustrate the significance of considering the WFP of MSW management practices. Because of the challenges in calculating a WFP, the following limitations are noted:

- The study relied on published values for raw materials and virgin-based manufacturing of recycled commodities, however the WFP of materials is expected to vary based on manufacturing technology, location, and climate. Furthermore, the WFP of some construction capital goods were not included due to data unavailability.
- Leachate generation rate and quality were based on the U.S. EPA MSW-DST assumptions; however leachate generation and quality are expected to vary based on waste characteristics, climate, top and bottom liner material and installation quality, liner degradation, and time frame used. Moreover, the leachate treatment efficiency is going to vary based on the implemented technology, which will also have an impact on the grey WFP of waste landfilling.
- Due to data limitations, the WFP of collection, processing, and remanufacturing of recyclables was not included in the WFP of recycled commodities; also, this study presented the WFP of a few recyclables commodities and should be expanded.

4.5. Conclusions

The WFP can be used as a tool to assess MSW management practices, energy products, and recycled commodities, however many data gaps exist. Regional variability has significant impact on the WFP (e.g. precipitation, energy mix offsets, cooling technology, generation efficiency, regulations, leachate management, manufacturing technology). Overall, the WFP has similar conclusions regarding relative impacts of waste management approaches to GHG (i.e., net WFP of landfills was greater than waste combustion due to energy benefits), however landfills have lower WFP compared to oil and several renewables with respect to electricity generation WFP. The recycling of renewable commodities (e.g., paper) had the largest WFP offsets. In future works, the WFP should be used as a tool to assess alternative MSW management practices (e.g., bioconversion of waste to biofuels such as diesel and ethanol).

CHAPTER 5: MULTI-LEVEL MULTI-CRITERIA ANALYSIS OF ALTERNATIVE FUELS FOR WASTE COLLECTION VEHICLES IN THE UNITED STATES

5.1. Introduction

The waste collection industry is driven by the need to reduce costs and emissions while increasing operation efficiency. These challenges encourage the collection industry to explore alternative fuel technologies including compressed natural gas (CNG); liquefied natural gas (LNG); biodiesel (B20, B100), and hydraulic-hybrid (an alternative to conventional diesel trucks, where trucks are able to recapture, store, and reuse braking energy (Bender et al., 2014). In fact, prior to 2009, diesel-fueled waste collection vehicles (WCVs) were the backbone of the U.S. waste collection industry with less than one percent of WCVs using alternative fuel (Rogoff et al., 2009). The recent relatively low prices of natural gas compared to high diesel prices have incentivized the industry in natural gas as an alternative fuel for their fleets. In 2012, Waste Management Inc., based in Houston, Texas, and a leading provider of comprehensive waste management services in North America, operated the largest natural gas collection vehicles fleet in North America with nearly 1,700 CNG and LNG vehicles. In the next five years, it is anticipated that 80% of Waste Management annual new trucks purchased will be fueled by natural gas. The company added 13 CNG fueling stations in the first-half of 2012, which brought their total to 31. Moreover, Waste Management planned to construct another 17 stations by the end of 2012 (Waste Management Inc., 2012). The second major waste hauler in the U.S., Republic Services, with currently more than 1,000 vehicles running on alternative fuels, plans to add 3,100 natural gas and other alternative-fueled WCVs by the end of 2015 (Republic Services, 2012). In 2012, WCV and transfer vehicles accounted for 11 percent of the total U.S. natural gas vehicles (NGVAMERICA, 2012).

In contrast, diesel fuel purchases were estimated to consume 7.5% of the industry revenues in 2012 (Smith, 2012).

Undoubtedly, fuel cost has been the driving factor for the waste industry. A comprehensive decision matrix that considers other factors such as changing policies, future fuel prices and uncertain fuel performance data, has not been developed. In the last three decades, the selection scheme for alternative fuels and energies has changed from a single-criterion cost-based assessment, to a multi-criteria analysis that considers environmental, social, operational, and political factors (Pohekar and Ramachandran, 2004; Cavallaro, 2005; Wang et al., 2009; Linkov and Moberg , 2011). Multi-criteria decision analysis (MCDA) methods have been used to rank alternative fuel buses for public transportation (Tzeng et al., 2005), alternative transportation fuels (Mohamadabadi et al., 2009), electricity generation alternatives (Cristóbal, 2011), municipal solid waste management alternatives (Herva and Roca, 2013), and landfill sites (Şener et al., 2006).

In a previous study, Maimoun (2011) presented the different factors associated with the selection criteria of alternative fuels for collection vehicles. A multifactorial assessed was presented by a heat map (a graphical representation which reflects better fuel performance by darker shades) (Maimoun, 2011). The use of MCDA methods allow decision makers to systematically select the best alternative with respect to selection criteria, while understanding the trade-off that occur in selecting different alternatives (Linkov and Moberg, 2011). In this study, MCDA methods will be used to rank alternative fuels for WCVs using a multi-level multi-criteria decision analysis framework (Read et al., 2013) that incorporates environmental and financial criteria, providing insights for better decision-making by the waste industry. Sensitivity analysis will be performed to determine the robustness of fuel rankings to changing policies, selection criteria, and fuel performance data.

5.2. Methods

Alternative fuels were identified based on a literature review. A fuel selection criteria that consider environmental and financial factors were established. The fuel performance data (a quantitative measure of the fuel performance with respect to each selection criteria) were obtained from the literature. Finally, two MCDA methods, Simple Additive Weighting (SAW) (Churchman and Ackoff, 1954) and (2) Technique for Order Preference by Similarity to Ideal Solution (TOPSIS) (Hwang and Yoon, 1981), were used to rank fuel alternatives for the waste collection industry using the multi-level environmental and multi-criteria approach (Read et al., 2013). The selection of these two methods was based on their ability to handle multi-attribute decision making problems. The following sections provide more details about the decision analysis process.

5.2.1. Fuel Alternatives for Waste Collection Vehicles

Nine different fuels could be considered for WCVs; gasoline, diesel, natural gas (Gordon et al., 2003), biodiesel (López et al., 2009), liquefied petroleum gas, hydraulic-hybrid (de Oliveira et al., 2014), hydrogen gas (FAUN, 2011), ethanol E85, and dimethyl ether (DME) (Tsuchiya and Sato, 2006). Only four fuel technologies were commercially available for WCVs, diesel, natural gas, biodiesel, and hydraulic-hybrid. Diesel-fueled WCVs can operate on fossil diesel or biodiesel (BD) blends (BD20 and BD 100), but may require engine modifications when using biodiesel blends (U.S. EIA, 2015a). BD100 is made of 100% biodiesel, while BD20 is a blend of 20% biodiesel and 80% fossil diesel (U.S. EIA, 2015a). In the U.S., biodiesel is produced from a diverse biomass feedstock, led by soybean oil which accounted for more than 50% in 2013 (U.S. EIA, 2015b). In this study, two sources of biodiesel were investigated; soybean as a primary source of biodiesel in the US, and algaculture as an alternative future source. Natural gas WCVs can operate either using CNG or LNG, and can be obtained from a fossil or biogenic source. In this study, fossil sources were categorized as North American or Non-North-American. Landfill gas (LFG) sourced natural gas was the only biogenic natural gas source considered in this study. LFG is comprised of mainly methane (50-60%) and carbon dioxide (40-40%) (Shin et al., 2005; U.S. EPA, 2012b). It also consists of hundreds of other compounds at lower concentrations such as oxygen, nitrogen, sulfur compounds, water vapor and organic compounds (U.S. EPA, 2000; Shin et al., 2005). In order to use LFG as an alternative vehicular fuel, LFG should be converted to pipeline quality natural gas, with high BTU content, through the separation of methane from carbon dioxide and other constituents (Hesson, 2008; U.S. EPA, 2000).

Accordingly, twelve alternative fuels or fuel blends were considered for the WCVs in the U.S. based on fuel type and source; (1) diesel, (2) CNG (North American), (3) CNG (Non-North American), (4) LNG (North American), (5) LNG (Non-North American), (6) hydraulic-hybrid, (7) CNG (LFG sourced), (8) LNG (LFG sourced), (9) BD20 (Algaculture), (10) BD20 (soybean), (11) BD100 (Algaculture), and (12) BD100 (soybean).

5.2.2. Fuel Evaluation Criteria

First, a multi-level fuel selection criteria matrix that considers environmental and financial factors was established (Figure 5.1). The upper level criteria were then broken down into sub-criterion categories, e.g. tail-pipe emissions (second level environmental criteria) of WCVs were evaluated based on carbon monoxide, carbon dioxide, nitrogen oxides, particulate matters, and total hydrocarbons emissions. Fuel performance data were presented for each alternative with respect to the sub-criterion category, e.g., fuel performance data were presented for carbon monoxide, carbon dioxide, nitrogen oxides, particulate matters, and total hydrocarbons emissions (level 3).

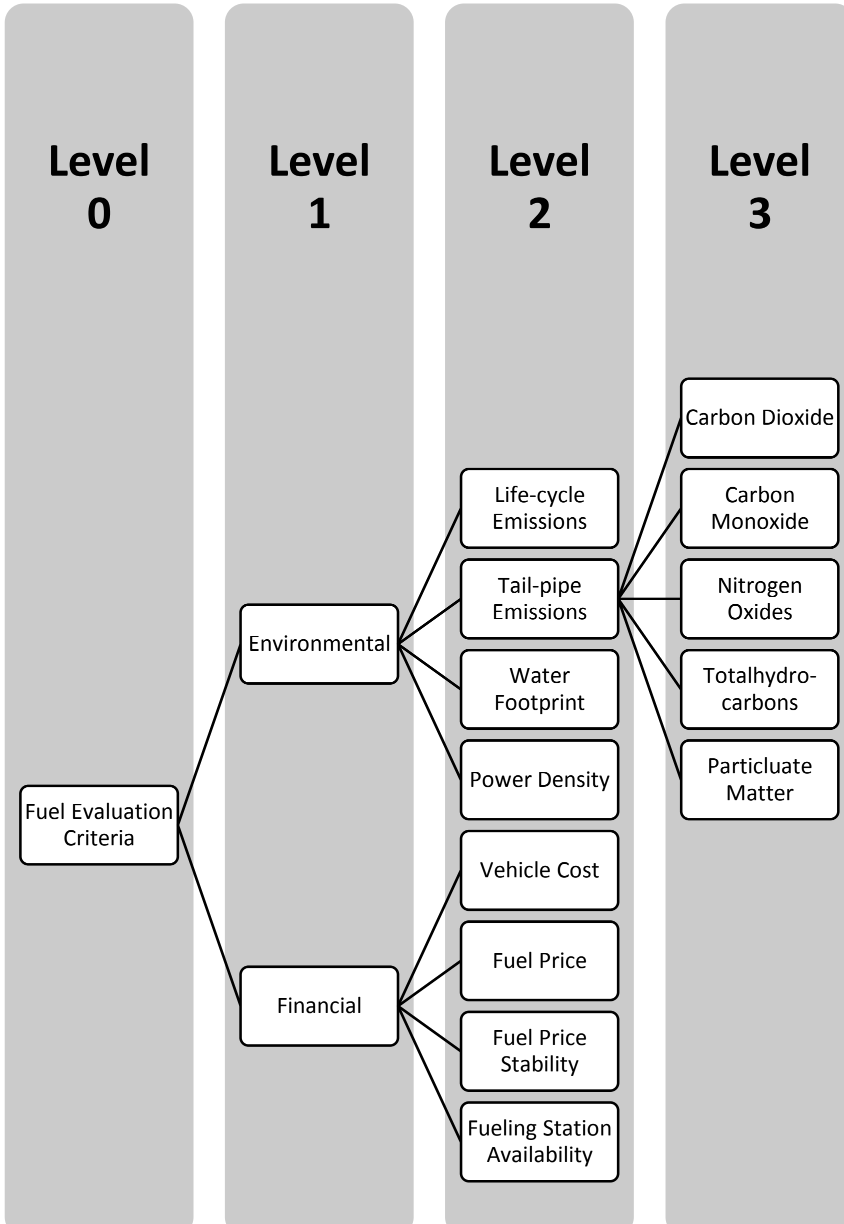

Figure 5.1: Multi-level multi-criteria decision making matrix.

5.2.2.1 Environmental Criteria

Four environmental criteria were considered in this study: life-cycle emissions of alternative fuels and fuel blends, tail-pipe emissions of alternative fuel WCVs, water footprint (WFP), and power density of alternative fuel and fuel blends.

5.2.2.1.1 Life-cycle Emissions of Alternative Fuels

Life-cycle emissions of alternative fuels and fuel blends had been calculated by Maimoun et al. (2013) using the greenhouse gases, regulated emissions, and energy use in transportation (GREET) Model provided by Argonne National Laboratory (U.S. DOE, 2012b). The life-cycle emissions associated with diesel, CNG (North American), CNG (Non-North American), LNG (North American), LNG (Non-North American), hydraulic-hybrid, LNG (LFG sourced), CNG (LFG sourced), BD100 (Algaculture), BD20 (Algaculture), BD100 (soybean), and BD20 (soybean) were estimated at 2.85, 3.01, 3.27, 3.14, 3.39, 2.33, 0.62, 0.5, 1.4, 2.52, 0.71 and 2.38 kg CO_{2eq} per collection vehicle kilometer travel (CVkmT), respectively (Maimoun et al., 2013).

5.2.2.1.2 Tail-pipe Emissions of Alternative Fuel WCVs

Tail-pipe emissions of WCVs include carbon dioxide (CO_2), carbon monoxide (CO), nitrogen oxides (NO_x), total hydrocarbons (THC) and particulate matter (PM). Tail-pipe emissions for conventional diesel-fueled WCVs were measured by Farzaneh et al. (2009) using two portable emissions measurement systems (PEMS). Emissions from conventional diesel-fueled WCVs were investigated under four different operation modes including (1) urban driving, (2) trash collection, (3) freeway driving, and (4) landfill activities (Farzaneh et al., 2009). For this study, a weighted average was calculated for each pollutant using the average emission factor associated with each driving mode and the fraction of the driving mode to the overall route. The average tail-pipe emissions from conventional diesel-fueled WCVs were estimated to be 2.8 kg/km, 17.1 g/km, 17.1 g/km, 0.6 g/km, and 0.06 g/km for CO_2, CO, NO_x, THC, and PM, respectively.

A different study by Texas Transportation Institute (2009) compared the tail-pipe emissions of CNG fueled WCVs relative to conventional diesel vehicle, e.g. the tail-pipe NOx emissions of CNG vehicles was found to be 96% less than conventional diesel WCVs (Table 5.1).

Tail-pipe emissions for LNG were assumed to be equal to CNG based on their identical chemical properties. According to the U.S. Environmental Protection Agency (EPA), the use of hydraulic-hybrid diesel WCVs has a potential fuel savings of up to 30%. Therefore, tail-pipe emissions from hydraulic-hybrid WCVs were assumed to be 30% less than conventional diesel-fueled WCVs (Hall, 2010). de Oliveira et al. (2014) also reported 15 to 25% improvement in fuel economy of heavy-duty hydraulic-hybrid WCV compared to conventional diesel-fueled WCVs. Tail-pipe emissions for buses running on BD20 and BD100 showed lower emissions compared to diesel buses, except for NO_X emissions (U.S. EPA, 2002). Relative emissions values shown in Table 5.1 were applied to the weighted average of the conventional diesel tail-pipe emissions to estimate alternative-fueled WCVs tail-pipe emissions.

Table 5.1: Alternative-fueled waste collection vehicle (WCV) tail-pipe emissions relative to diesel-fueled vehicles.

Fuel Category	CO_2	CO	NO_x	THC	PM	Source	Assumption
CNG (Source: American, non-American, LFG)	-27%	+1,200%	-96%	5,700%	--	Texas Transportation Institute (2009)	
LNG (Source: American, non-American, LFG)	-27%	+1200%	-96%	5,700%	--	Texas Transportation Institute (2009)	Tail-pipe emissions from LNG equal to CNG
hydraulic-hybrid	-30%	-30%	-30%	-30%	-30%	Hall (2010)	Hybrid waste collection vehicles with 30% fuel saving will have 30% less tail-pipe emissions
BD20 (Source: Algaculture, soybean)	--	-11%	+2%	-21%	-10%	U.S. EPA (2002)	Similarity between Waste Collection Vehicles and Heavy-duty Vehicles.
BD100 (Source: Algaculture, soybean)	--	-47%	+10%	-68%	-45%		

5.2.2.1.3 Water Footprint (WFP) of Alternative Fuels and Fuel Blends

The WFP is a measure of both the direct and indirect use of fresh water over the entire process life cycle (Hoekstra et al., 2009). It consists of three components; blue accounting for the consumption of surface and groundwater resources; green referring to consumption of rainwater stored in the soil as soil moisture, normally lost through evapotranspiration; and grey, relating to water pollution and defined as the volume of freshwater that is required to dilute pollutants to meet existing water quality standards (Hoekstra et al., 2009). The total WFP of any process, product, or energy source is the summation of the blue, green and grey WPFs. The total WFP associated with alternative fuels was obtained from the literature (Gerben-Leenes et al., 2008; Singh et al., 2011), except for LFG (Table 5.2). The WFP of LFG source vehicular fuel was not evaluated previously; so the WFP of LFG conversion to vehicular fuel was calculated and is presented in this section.

Currently, commercial methods available to purify LFG include: (1) water scrubbers, (2) gas cooling separation, and (3) membrane separation (Läntelä et al., 2012). In this study, the WFP of LFG conversion to vehicular fuel was calculated for a water scrubber with water recycling to remove carbon dioxide, as it is considered the most cost effective and widely use technology for upgrading LFG to vehicular fuel (Hunter and Oyama, 2000; Rasi at al., 2008). The process equipment consists of absorption and desorption columns, pumps, compressors, and drying unit (Läntelä et al., 2012).

In order to calculate the WFP of LFG conversion, it was necessary to set the system boundaries of the process. The function of any landfill is the disposal of municipal solid waste and LFG is a byproduct of waste landfilling. According to the U.S. EPA (2012d), large landfills are required to collect LFG for beneficial use or flaring. As a result, the system boundaries for calculating the WFP of LFG conversion to vehicular fuel excluded landfill construction and operation, LFG collection, and any condensate generated in the process, and only includes (1) water evaporated during processing and need to be replaced, (2) electricity consumption, and (3) any WFP offset as a result of energy recovered. The functional unit used was cubic meters of water per GJ of vehicular fuel produced.

The energy content in standard cubic meter methane is 37,700 KJ/Nm^3. Therefore, the energy recovered in converting a standard cubic meter of LFG, assuming that 100% of the methane in LFG is recovered, is equal to 18,900-22,600 KJ per Nm^3 of LFG. The WFP of fossil natural gas is 110 L per GJ (Gerbens-Leenes et al., 2008), therefore, a WFP offset between -2.1 and -2.5 L per Nm^3 of LFG converted is associated with energy recovery from LFG.

In a pilot study described by Rasi et al. (2008) and Läntelä et al. (2012) to convert 7.41 Nm^3/h of LFG to vehicular fuel using water scrubbers with complete water recycling, Läntelä et al. (2012) estimated that about 1% of circulating water (700 l in total) was evaporated or lost during the upgrade process (3–6 h). Therefore, it is estimated that the process WFP for water replacement is 0.21 L per Nm^3 LFG processed. The upgrade process electricity consumption was estimated by Läntelä et al. (2012) to be between 0.43-0.55 kwh/Nm^3. The WFP of the US electricity was estimated for the 2013 U.S. electric grid energy mix using the WFP of different energy sources compiled by Hadian and Madani (2013). The overall total WFP of the U.S. electric energy mix was calculated at 9 L per kWh, therefore it is estimated that the WFP associated with energy consumption is between 3.9 and 4.95 L per Nm^3.

The total WFP of converting LFG to vehicular is estimated to be between 4.1 and 5.2 L per Nm^3 of LFG processed, while the net (accounting for offset) is estimated between 1.6 and 3.1 L per Nm^3 or between 0.07 and 0.16 m^3 per GJ of vehicular fuel. The net WFP of LFG sourced vehicular fuel is impacted by the high WFP of the U.S. electric grid and relatively low WFP of fossil natural gas. In comparison, the WFP of LFG-sourced natural gas is comparable to the WFP of fossil natural gas, depending on the quality of LFG. Also, the process WFP calculations are based on a pilot study and it was assumed that the data would scale to commercial facilities. This estimate does not include the WFP of potential contamination or water discharge to the surroundings in case of failure of the water recycling system, or initial construction material, e.g. absorption and desorption columns, where no documentation was found. On average, natural gas has the lowest WFP, followed by LFG-sourced natural gas, while diesel has an average to moderate WFP. The production of biodiesel was found to have the highest WFP, associated with growing and processing of energy crops. The WFP data are presented in Table 5.2.

Table 5.2: WFP and Power Density of Alternative Fuels and Fuel Blends.

	WFP			Power Density		
	M^3/GJ	Source	Assumptions	W/m^2	Source	Assumptions
Diesel	1.06	Gerbens-Leenes et al. (2008)	WFP of diesel equals crude oil extraction and processing	10^3 to 10^4	Smil (2010)	The Power Density of Oil Field was used.
CNG (Fossil Source)	0.11	Gerbens-Leenes et al. (2008)	WFP of CNG equals natural gas extraction and processing	10^3 to 10^4	Smil (2010)	Assumed Similar to Diesel
CNG (LFG Source)	0.07-0.16	This Study		10	Amini and Reinhart (2011)	
LNG (Fossil Source)	0.11	Gerbens-Leenes et al. (2008)	Liquefaction of natural gas to LNG consumes water; so it was assumed to be at the high end of CNG WFP	10^3 to 10^4	Smil (2010)	Assumed Similar to Diesel
LNG (LFG Source)	0.07-0.16	This Study		10	Amini and Reinhart (2011)	
Biodiesel (Soybean)	383	Singh et al. (2011)	Average WFP of biodiesel	$1.32x10^{-5}$	Pienkos (2007)	
Biodiesel (Algaculture)	<379	Singh et al. (2011)	Average WFP of biodiesel	$3.3x10^{-4}$ to $2.75x10^{-3}$	Pienkos (2007)	

5.2.2.1.4. Power Density of Alternative Fuels and Fuel Blends

The power density (also called production density or energy footprint) illustrates the footprint needed to produce one unit of energy (Smil, 2010), and it is essential to estimate the land footprint of energy sources. The power density was represented by Watts generated per area of land (m^2). Biodiesel production, either from algaculture or soybeans, was found to have a low power density compared to all other alternative fuels. The power density associated with fuel production are listed in Table 5.2.

5.2.2.2. Financial Criteria

In this study, four financial criteria were considered; vehicle cost, fuel cost, fuel price stability and fueling station availability. A quantitative measure of each alternative with respect to each criteria will be presented in this section.

5.2.2.2.1. Vehicle Cost of Alternative Fuel Vehicles

Vehicle cost is a significant part of the capital cost associated with switching to an alternative fuel, therefore it was considered in the selection criteria. The average vehicle cost was reported for each alternative in U.S. dollars per WCV (Table 5.3).

5.2.2.2.2. Fuel Cost

The relatively low priced natural gas compared to diesel shaped the recent history of vehicle purchases by the waste collection industry; this demonstrates the importance given to fuel prices. In order to estimate the fuel cost, the average fuel mileage was adopted from a previous study (Maimoun et al., 2013). The fuel mileage was used with the 2012 national average fuel (U.S. DOE, 2012c) to estimate the fuel cost in U.S. dollars per CVKmT.

5.2.2.2.3. Fuel Price Stability

Fuel price stability was considered a part of the financial criteria. The fuel price stability was measured by the standard deviation of the U.S. national fuel prices during 2012. The cost of conversion of LFG to vehicular fuel was assumed to be stable over the course of one year (2012).

5.2.2.2.4. Fueling Stations Availability

The limited number of CNG/LNG fueling stations forced waste haulers to invest in building new stations, while gradually switching new vehicles purchases to natural gas as the price of gas plummeted. This demonstrates the significance of fueling station availability to select among alternative fuels, therefore the number of commercially available fueling stations was used to quantify the infrastructure criterion for each alternative (Table 5.3). In the case of CNG/LNG from LFG, the number of US landfill to natural gas projects was used. In 2008, there were only 20 sites converting LFG to vehicular fuel; however there were more than 425 landfills in the US of which about 300 are used to generate electricity and 110 commercial/industrial heating fuel (Hesson, 2008), which shows the potential for more landfill sites that can be used to produce vehicular fuel.

Table 5.3: Financial Performance Data.

	Vehicle Cost		**Fuel Cost**				**Fuel Price Stability**		**Fueling Stations**	
	U.S. Dollar per Collection Vehicle	Source/ Assumption	Mileage (Km per L diesel equivalent)	Unit Price		Travel Cost (U.S. Dollar per CVKMT)	Standard Deviation of price (2012)	Source/ Assumption	No of Stations in the US	Source
Diesel	160,000-200,000	Gordon et al. (2003)	1.2	$1.09 per diesel Equivalent L	2012 National Average Price, U.S. DOE (2012c)	0.91	0.24	U. S. DOE, 2012c	128,887	U.S. DOE (2012d)
CNG	200,000-250,000	Gordon et al. (2003)	1.0	$0.613 per diesel Equivalent L	2012 National Average Price, U.S. DOE (2012c)	0.61	0.66	U.S. DOE, 2012c	1048	U.S. DOE (2012d)
LNG	200,000-250,000	Similar to CNG	0.95	$0.613 per diesel Equivalent L	LNG price Similar to CNG	0.65	0.66	U.S. DOE, 2012c	53	U.S. DOE (2012d)
Hydraulic Hybrid	260,000-300,000	Danna (2011)	1.5	$1.08 per diesel Equivalent L	2012 National Average Price, U.S. DOE (2012c)	0.81	0.24	U.S. DOE, 2012c	128,887	U.S. DOE (2012d)
CNG (Source: LFG)	200,000-250,000	Gordon et al. (2003)	1.0	$5 and 8 per MBtu (Average Price of $6.5 per MBtu was used)	Hesson (2008)	0.22	0	The price of LFG is assumed to be constant	20	Hesson (2008)
LNG (Source: LFG)	200,000-250,000	Similar to CNG	0.95		LNG price Similar to CNG	0.24	0	The price of LFG is assumed to be constant	20	Hesson (2008)
BD100	160,000-200,000	Similar to regular diesel	1.2	$1.26 per diesel Equivalent L	2012 National Average Price, U.S. DOE (2012c)	1.05	0.52	U.S. DOE, 2012c	621	U.S. DOE (2012d)
BD20	160,000-200,000	Similar to regular diesel	1.2	$1.12 per diesel Equivalent L	2012 National Average Price, U.S. DOE (2012c)	0.93	0.33	USDOE, 2012c	621	U.S. DOE (2012d)

5.2.2. MCDA methods

Two MCDA methods were used to rank alternative fuels with respect to the selected criteria; SAW (Churchman and Ackoff, 1954) and TOPSIS (Hwang and Yoon, 1981). The selection of these two methods was based on their ability to handle multi-attribute decision making problems. SAW (Churchman and Ackoff, 1954) is the most widely known MCDA method, based on comparing the weighted average of alternatives performance data with respect to a selection criteria (Afshari et al., 2010). On the other hand, TOPSIS is based on choosing a hypothetical ideal solution; the alternative that has the shortest geometric distance from the positive ideal optimal solution and the longest geometric distance from the negative solution is the optimal solution. TOPSIS can also accommodate different criteria weight in ranking alternatives (Hwang and Yoon, 1981).

SAW and TOPSIS require a comparable scale for all elements in the decision matrix, therefore performance values were normalized with respect to each criterion. The normalized performance criteria values were obtained for beneficial criteria (larger utility, greater performance) using Equation 5.1 (Nguyen and Gordon-Brown, 2012).

$$r_{ij} = \frac{x_{ij} - min_j}{max_j - min_j} \qquad (5.1)$$

Where:

r_{ij} = Normalized value of alternative (i) with respect to criteria (j) (0-1)

x_{ij} = Performance value of alternative (i) with respect to criteria (j)

max_j = Maximum performance value with respect to criteria (j)

min_j = Minimum performance value with respect to criteria (j)

For cost criteria (the smaller the rating, the better the preference), the normalized value was calculated using Equation 5.2 (Nguyen and Gordon-Brown, 2012).

$$r_{ij} = \frac{max_j - x_{ij}}{max_j - min_j} \qquad (5.2)$$

5.2.2.1. Simple Additive Weighting (SAW)

The SAW method (Churchman and Ackoff, 1954) compares alternatives using the comparison index (SAW_j) calculated in Equation 5.3. The higher the index value the better the performance.

$$SAW_j = \sum_{j=1}^{n} W_j * r_{ij} \tag{5.3}$$

Where:

W_j = Entropic weight of each criterion j

The entropic weight (W_j) of each criterion (j) is used to determine the weight of each criterion based on the dispersion of the performance values (Chan et al., 1999). W_j of each criterion can be calculated using Equation 5.4 as described by Madani et al. (2014).

$$W_j = \frac{d_j}{\sum_{j=1}^{n} d_j} \tag{5.4}$$

Where:

$$d_j = 1 - E_j$$

E_j is the entropy of normalized performances under a given criterion and can be calculated using Equation 5.5 as described by Madani et al. (2014).

$$E_j = -k \sum_{i=1}^{m} P_{ij} \,.\ln(P_{ij}) \tag{5.5}$$

Where:

m = Total number of alternatives

$$k = \frac{1}{\ln(m)}$$

$$P_{ij} = \frac{r_{ij}}{\sum_{i=1}^{m} r_{ij}}$$

5.2.2.2. Technique for Order Performance by Similarity to an Ideal Solution (TOPSIS)

The TOPSIS method (Hwang and Yoon, 1987) selects the alternative that has the minimum relative performance distance from an ideal solution. The relative distance (CL_i^+) of each alternative to the ideal solution is calculated using Equation 5.6 as described by Madani et al. (2014).

$$CL_i^{+} = \frac{d_i^{+}}{d_i^{+} + d_i^{-}} \qquad (5.6)$$

Where:

$$d_i^{+} = \left[\sum_{j=1}^{n} \left(V_{ij} - V_j^{+}\right)^2\right]^{0.5}$$

$$d_i^{-} = \left[\sum_{j=1}^{n} \left(V_{ij} - V_j^{-}\right)^2\right]^{0.5}$$

The normalized utility (N_{ij}) is used to calculate the weighted normalized performance (V_{ij}) of each alternative under each criterion using equation 5.7. The best (V_j^+) and the worst (V_j^-) performance of the alternatives under each criterion are determined, and used to calculate the distance of each alternative from the best and the worst scenario as shown previously in Equation 5.6.

$$V_{ij} = N_{ij}.W_{ij} \qquad (5.7)$$

Where:

$$N_{ij} = \frac{r_{ij}}{\sqrt{\sum_{i=1}^{m} r_{ij}^2}}$$

At every level of the decision matrix, SAW and TOPSIS were used to calculate the comparison indices and relative distances of each alternative. The comparison indices (or relative distances) were normalized using Equations 5.1 and 5.2, and used as a performance input value for the upper level.

5.3. Results and discussion

TOPSIS and SAW were used to rank fuel alternatives for the waste collection industry with respect to the multi-level environmental and financial decision matrix, Figure 5.2 and 5.3. The overall ranking placed conventional diesel-fueled WCVs as the best option under the decision matrix, followed by hydraulic-hybrid, LFG-sourced natural gas, North American and non-North American natural gas, and biodiesel fuels. The results of the two methods were consistent. Environmentally, WCVs fueled with fossil fuels (diesel and natural) were closer to the ideal than biogenic fuels (BD and LFG); the inclusion of the WFP and power density as environmental measures placed biogenic fuels, biodiesel and LFG, far from being the ideal fuel option. Environmentally, CNG and LNG WCVs fueled by American fossil natural gas had slight advantages over WCVs fueled with non-American natural gas or diesel. Hydraulic-hybrid WCVs were the closest to the optimal solution with respect to the environmental criteria, because fuel savings compared to diesel placed it closer to the optimal environmental option than diesel. Financially, diesel and hydraulic-hybrid ranked closest to the ideal solution under the decision matrix. The vehicle cost of hydraulic-hybrid vehicles averaged $100,000 more than conventional diesel-fueled WCVs; however the fuel savings associated with hydraulic hybrid WCVs placed it at a similar distance from the ideal solution as conventional diesel-fueled WCVs. Natural gas (CNG and LNG) and biodiesel were affected by the current lack of fueling stations. The fuel price of biodiesel placed this option far from the ideal solution as it is currently the most expensive alternative. LFG has the cheapest price; however the availability of LFG fueling station impacted the financial and overall performance of this alternative.

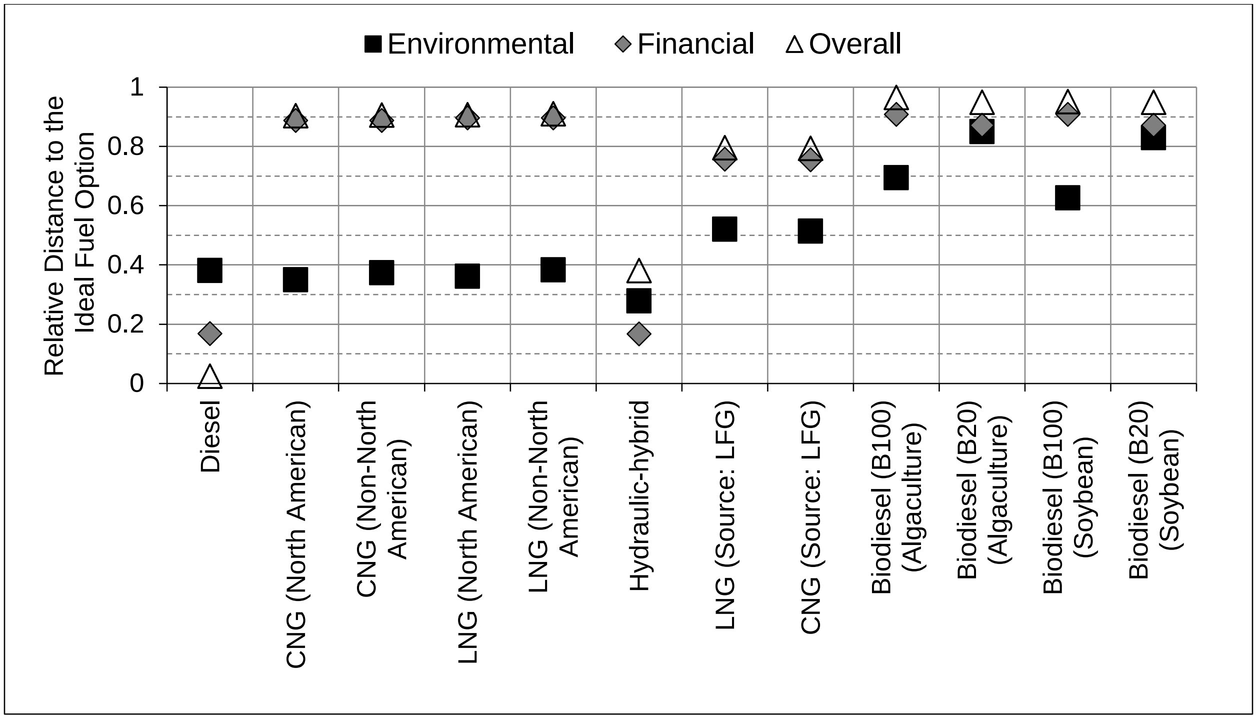

Figure 5.2: Relative Distances (TOPSIS Analysis) of Fuel Options from the Ideal Option

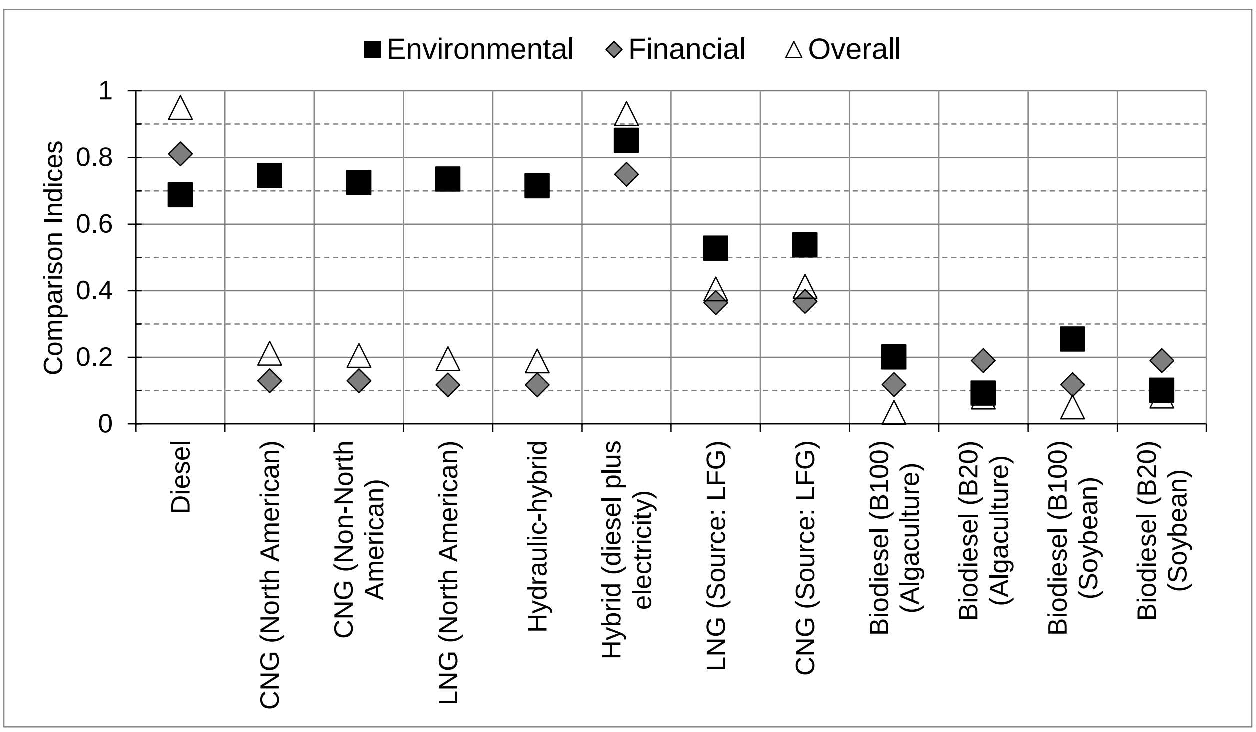

Figure 5.3: Comparison Indices (SAW Analysis) of Fuel Options

5.3.1. Significance of the Selection Criteria

In the previous analysis, fuel rankings were based on the selected decision matrix; however it is imperative to assess how sensitive the fuel rankings are to the selection criteria considered by DMs. Therefore, an analysis was conducted by eliminating one or two criteria from the decision matrix, then determining the relative distance of alternatives to the ideal solution (TOPSIS analysis). The following five scenarios were considered:

Scenario 1: Eliminate the water footprint criterion,

Scenario 2: Eliminate the WFP and the power density criteria,

Scenario 3: Eliminate the fueling station criterion,

Scenario 4: Eliminate the fuel price stability criterion,

Scenario 5: Eliminate the fueling station and fuel price stability criteria.

The five scenario results are illustrated in Figure 5.4. For comparison, the results from the previous analysis using the complete decision matrix were labeled **Scenario 0**. In **Scenario 1**, the elimination of the WFP criterion from the decision matrix did not impact the environmental or overall fuel ranking because the ranking of agricultural-based fuel alternatives was also affected by low power density as compared to fossil fuels. Alternative fuels with high WFP are associated with low power density, as a result, the elimination of the WFP alone did not affect the environmental or overall ranking of agriculture-based fuel alternatives. In **Scenario 2**, the elimination of the WFP and the power density from the decision matrix changed the environmental ranking of fuel alternatives so that biofuels (LFG-sourced natural gas and biodiesel) ranked ahead of fossil fuels. Biogenic fuels were considered the best based on life-cycle emissions and some tail-pipe pollutants, however they are associated with high WFP and low power density. LFG-sourced natural gas ranked as the best alternative followed by BD100 (soybean then algaculture), the overall ranking of alternatives was slightly affected by removing the WFP and power density criteria from the decision matrix, as LFG-source natural gas ranked third after conventional diesel and hydraulic-hybrid. Biodiesel has favorable life-cycle emissions however its production is associated with high WFP and low power density. These results signify the importance of considering the WFP and power density criteria as environmental measures in addition to traditional life-cycle and tail-pipe emissions. It also suggests the use of different feedstock (e.g. waste) for the production of biodiesel, which might reduce the WFP and the power density of biodiesel production, making it more favorable.

In **Scenario 3**, the fueling station criterion was eliminated from the decision matrix and LFG-sourced natural gas ranked as the best alternative from a financial prospective. Diesel and hydraulic-hybrid were ranked next, followed by BD20, North American, non-North American natural gas, BD100. Therefore, LFG-sourced natural gas is considered the best option for WCVs when available. In **Scenario 4**, the fuel price stability was eliminated from the decision matrix makes diesel and hydraulic-hybrid the optimal financial solution followed by LFG-sourced natural gas, however the overall ranking did not change significantly from Scenario 0. In **Scenario 5**, the elimination of fueling station and fuel price stability criteria ranked LFG-sourced natural gas as the best financial alternative followed by North American fossil natural gas. This scenario was found to represent the status-quo of the waste collection industry as the industry is leaning toward fossil natural gas, driven by low natural gas prices. In the next section, a systematic sensitivity analysis of the fuel ranking to instability of fuel prices was evaluated for the status-quo scenario.

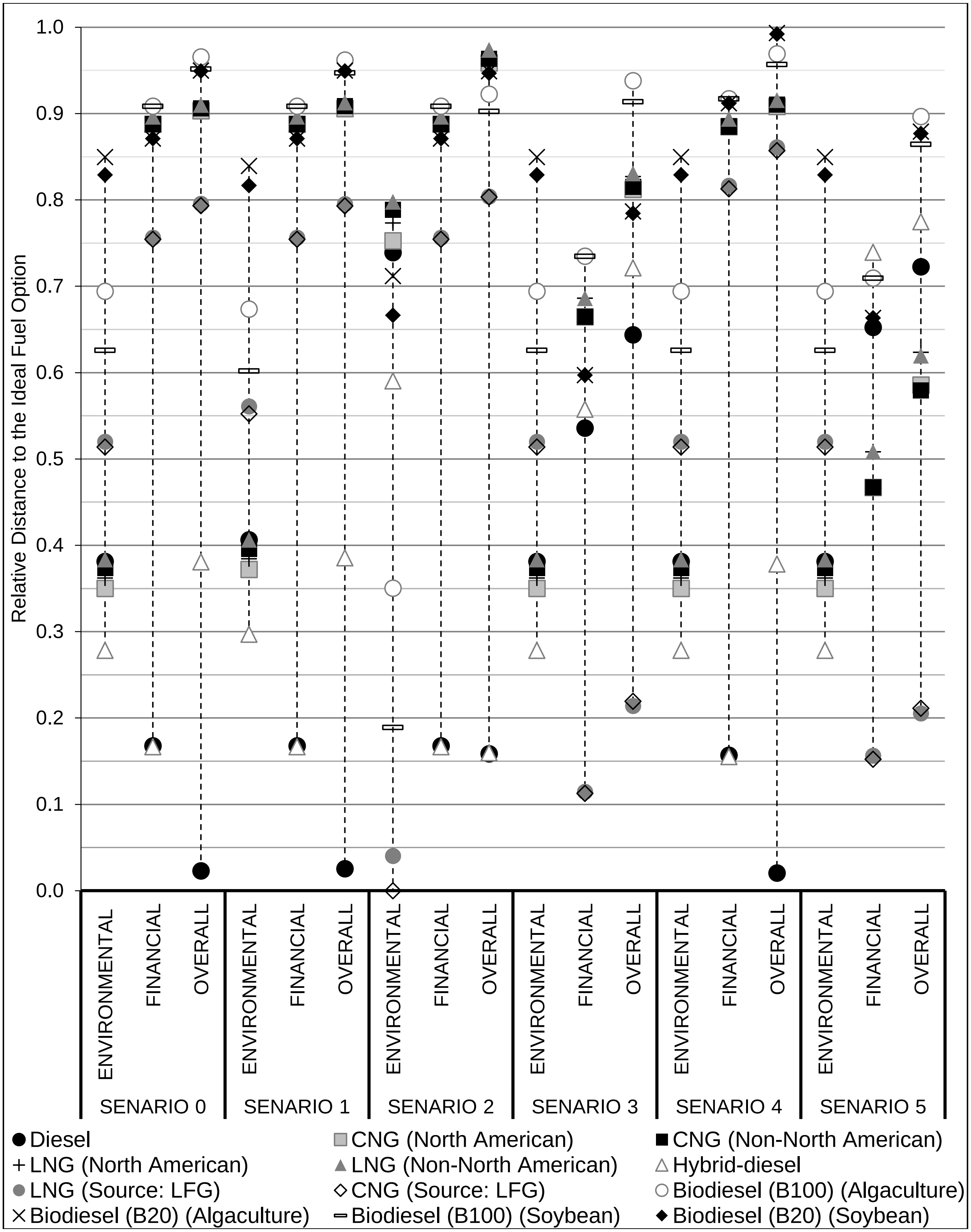

Figure 5.4: Significance of the Selection Criteria.

5.3.2. Systematic Sensitivity Analysis of Alternative Fuel Price

A systematic sensitivity analysis of alternative fuel price was conducted by evaluating the relative distances (TOPSIS) of each alternative from the ideal financial fuel option (Figure 5.5) and ideal overall fuel option (Figure 5.6), using five different price scenarios for diesel, natural gas, LFG, and biodiesel. In the analysis, the relative distances were calculated for each alternative while varying the fuel price of each alternative by -50%, -25%, +25%, and +50% of the current fuel price. The fueling station and fuel price stability criteria were eliminated from the decision matrix during the analysis to ensure the status-quo scenario was determined by the sensitivity analysis. The financial criteria consisted of the vehicle cost and fuel price, while environmental criteria included life-cycle emissions, tail-pipe emissions, WFP, and power density. The low number of fueling station gave advantage to some alternatives over others, while the fuel price stability criterion was excluded as the analysis gauges the sensitivity of ranking to changing fuel prices. The purpose of this analysis was to determine how sensitive the fuel ranking is to changing fuel price, as the industry builds more natural gas fueling stations based on the current natural gas prices.

Financially, CNG and LNG collection vehicles fueled with LFG-sourced natural gas ranked as the best alternatives. However, it was noticed that a 50% decrease in diesel fuel price placed diesel in the same rank as LFG-sourced natural gas. Also, a 50% decrease in fossil natural gas prices moved fossil CNG and LNG closer to LFG-sourced natural gas; however the LFG-sourced natural gas continued to rank as the best alternative. The ranking of diesel and hydraulic-hybrid was found to be more sensitive to fuel price, a drop of diesel price by 25% ranked diesel better than natural gas, while a 50% drop ranked hydraulic-hybrid as favorable as fossil natural gas. On the other hand, a 25% increase in diesel price ranked diesel and hydraulic-hybrid behind all other alternatives. Fossil CNG and LNG ranked behind LFG-sourced natural gas, however any increase in natural gas prices moved the alternative away from the ideal solution and in the case of a 50% increase, fossil natural gas ranked behind diesel and hydraulic-hybrid. LFG-sourced natural gas continued to rank as the best alternative even at a 50% increase in fuel price. Finally, BD20 and BD100 rankings are sensitive to changing fuel price. A 50% decrease in biodiesel price ranked BD20 and BD100 second after LFG-sourced natural gas, while a 25% ranked BD20 in between fossil CNG and LNG. An increase in biodiesel prices moved diesel toward fossil natural gas, a result of dispersion of fuel prices as biodiesel prices currently are highest.

Overall, LFG-sourced natural gas continued to rank as the best alternative with respect to the overall environmental and financial criteria, except at a 50% decrease in diesel prices (Figure 5.6). CNG and LNG collection vehicles fueled with North-American natural gas ranked second after LFG-sourced natural gas, however any increase in prices could move diesel and hydraulic-hybrid ahead of fossil natural gas (North American or non-North American). Fossil natural gas continued to rank as the second alternative after LFG-sourced natural gas except when natural gas prices were increased by 50% or diesel prices dropped by 25 to 50%. The overall ranking of LFG-sourced natural gas, BD 20, or BD100 was not significantly affected by changing fuel prices.

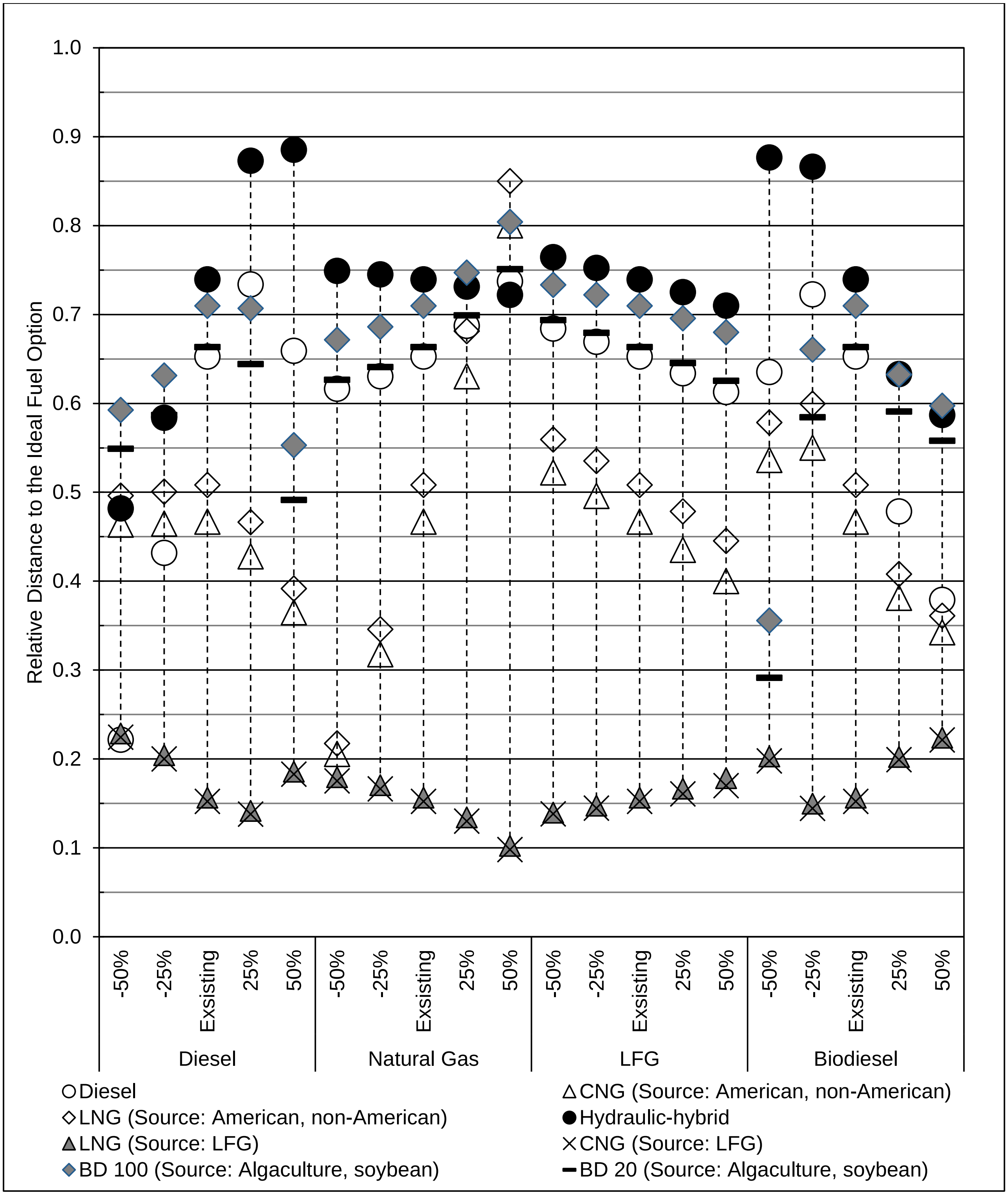

Figure 5.5: Systematic Sensitivity Analysis of the Financial Performance. (Relative distances (TOPSIS analysis) were calculated for each fuel using five different fuel pricing for each alternative; -50% of the current fuel price, -25% of the current fuel price, existing, +25% of the current fuel price, and +50% of the current fuel price).

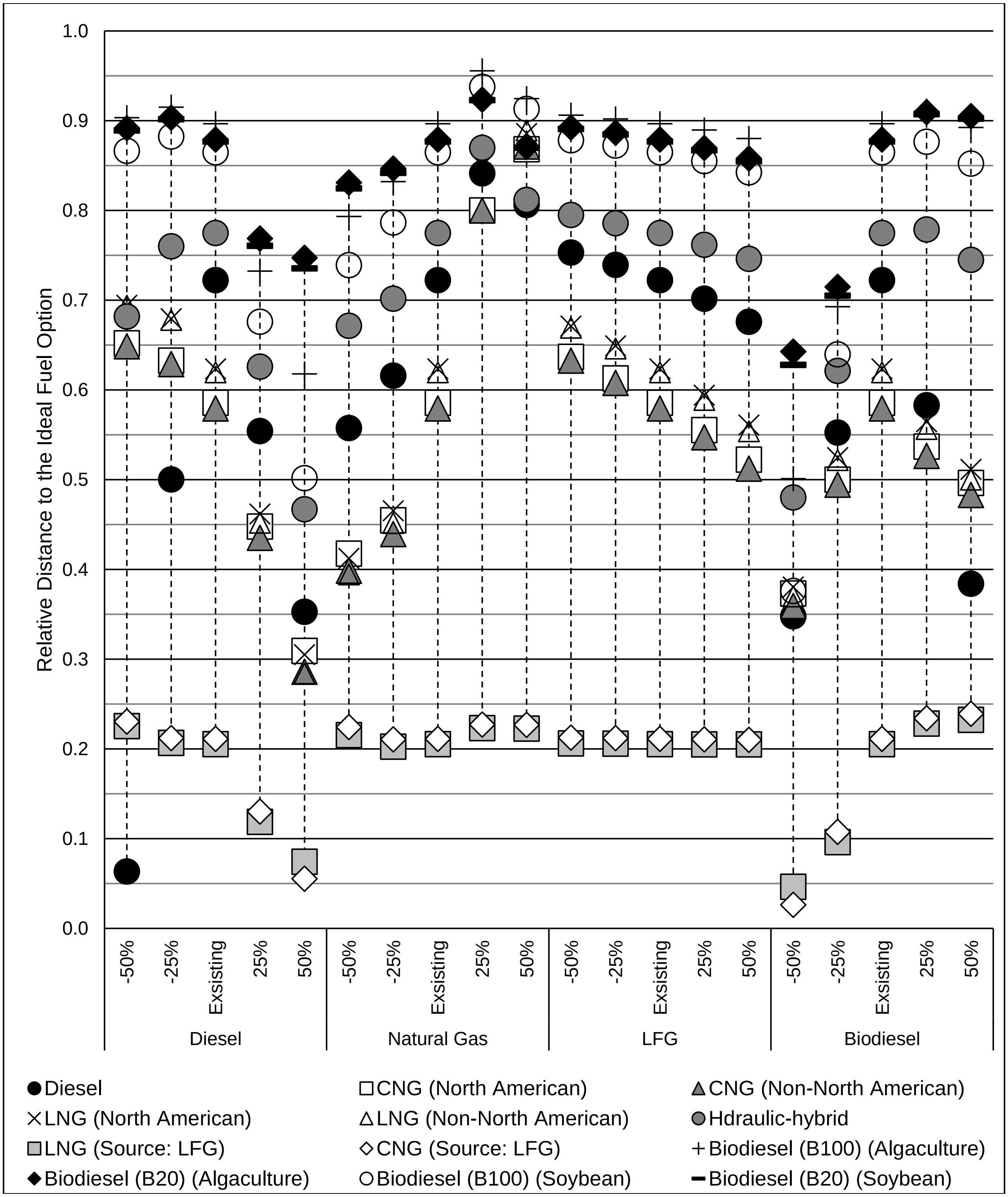

Figure 5.6: Systematic Sensitivity Analysis of the Overall Performance. (Relative distances (TOPSIS analysis) were calculated for each fuel using five different fuel pricing for each alternative; -50% of the current fuel price, -25% of the current fuel price, existing, +25% of the current fuel price, and +50% of the current fuel price).

5.4. Conclusions

MCDA tools were used to rank fuel alternatives for the waste collection industry with respect to a multi-level environmental and financial decision matrix. The environmental criteria consisted of life-cycle emissions, tail-pipe emissions, water footprint, and power density, while the financial criteria comprised of vehicle cost, fuel price, fuel price stability, and fueling station availability. Environmentally, hydraulic-hybrid and fossil natural gas, performed better than conventional-diesel; however, the vehicle cost of hydraulic-hybrid and lack of fueling stations for natural gas affected the financial ranking, although fuel price savings were observed for both options. The overall analysis using the environmental and financial criteria showed that conventional-diesel, followed by hydraulic-hybrid WCVs are the best alternatives, followed by LFG-sourced natural gas, fossil natural gas, and biodiesel. This fuel ranking changed as different decision matrices were used; signifying the importance of the selection criterion considered by decision makers. The elimination of the water footprint and power density criteria from the environmental criteria ranked biodiesel 100 (BD100) as an environmentally friendly alternative compared to other fossil fuels (diesel and natural gas). This result signifies the importance of considering WFP and power density criteria as environmental measures in addition to traditional life-cycle and tail-pipe emissions. The elimination of the fueling station criterion from the financial decision level ranked landfill gas (LFG) sourced natural gas as the best option; suggesting that LFG-sourced natural gas is the best alternative to fuel WCV when accessible. The elimination of the fueling station criterion and fuel price stability criterion from the decision matrix ranked fossil natural gas second after LFG-sourced natural gas. This scenario characterizes the status-quo of the industry; the waste collection industry is driven by low natural gas prices compared to other alternatives, and has set investment plans to build natural gas fueling stations. A systematic sensitivity analysis was used to determine the impact of changing fuel prices on decisions. The financial ranking of all alternatives, except LFG-sourced natural gas, was found to be sensitive to changing fuel prices. The overall ranking of diesel and natural gas was found to be more sensitive to changing fuel price as compared to LFG-sourced natural gas, BD20 or BD100.

CHAPTER 6: CONCLUSIONS AND RECOMMENDATIONS

Every municipality faces the need to optimize residential curbside collection (RCC) programs, in an attempt to reduce collection emissions, and increase waste diversion. Simultaneously, they are challenged by the need to select the best management practice and collection fuel that minimize environmental impacts while reduce cost. The goal of this study was to select the best RCC program, management practice, and collection fuel.

First, the study determined the effect of RCC system design on waste generation rates and recycling efficiency, which in turn affects waste management cost and environmental impacts. The results showed that the design of RCC programs (i.e. collection frequency and collection system) can significantly impact garbage and recyclables generation rates, and consequently determine environmental and economic impacts of collection systems. Residents' participation rate in curbside recycling was found to be an essential factor in determining the overall environmental performance of collection programs.

RCC programs are a part of the overall municipal solid waste (MSW) management system that manages, treats and disposes waste. The impact of MSW management practices on water resources has not been assessed yet. Therefore, this study used the water footprint methodology (the indicator of life-cycle impact of a process on water resources (Hoekstra et al., 2009)) to explore the impact of MSW management practices on water resources. Evaluating the WFP of MSW management practices is vital to a better understanding of the trade-offs between water use efficiency and other environmental burdens (e.g. GHG emissions). The WFP will be helpful to managers selecting the appropriate MSW treatment and disposal approaches in different locations of the U.S. with respect to water availability and impacts, particularly in water scarce areas. Moreover, the WFP will be important to compare the WFP of MSW management practices byproducts (e.g. electricity and recyclables) and other conventional sources.

Finally, collection fuel is a major operational element in RCC programs and MSW management practices. As the waste industry moves away from fossil fuels, the selection of an alternative collection fuel is associated with environmental and financial criteria. Multi-criteria decision making (MCDM) tools were used to rank alternative fuels for waste collection vehicles. A sensitivity analysis was used to determine the robustness of the ranking to changing selection criteria and performance data. This analysis can be used by the waste industry to select the best alternative fuel with respect to a selection criteria.

6.1. Policy Change Recommendations

An optimal design of RCC programs and a better recycling participation by residents will have a positive impact toward achieving Florida's recycling goal of 75% waste diversion. The implementation of a single-stream (SS) recycling system improved recycling rates, while it also reduced garbage collection rates. The study findings supported the current trends in switching to SS recycling systems combined with larger recycling containers. The study also found that recycling participation rate has a significant influence on the overall environmental performance of RCC programs, therefore municipalities are encouraged to monitor recycling participating rates and implement programs to improve it.

The study found that 30% of Florida RCC programs are providing two days of garbage collection. Reducing garbage collection frequency had positive environmental and economic effects, however it is usually opposed by Florida homeowners. As an alternative plan, municipalities are encouraged to split the two days of garbage collection service to separate collection lines of garbage and food-waste. This will help divert food waste from landfills, reducing early landfill gas emissions, while having a minimal effect on the potential recovered energy (Amini, 2011).

The fate of residential waste is usually determined by the RCC programs (Figure 1.1). The design of RCC programs is intended to achieve maximum waste diversion through recycling. Recycling is the best management practices as it offset emissions, while protecting resources through replacing virgin material manufacturing. As for the rest of the MSW stream (garbage stream), decision-makers are confronted by the need to select a management practice based on environmental and financial performance. In the literature, MSW management practices were compared based on GHG emissions; air, water, and soil releases; and cost. The impact of MSW management practices on water resources has been neglected, thus this study aimed to incorporate the WFP in the decision-making process. The WFP will be helpful to managers selecting the appropriate MSW treatment and disposal approaches in different locations of the U.S. with respect to water availability and impacts, particularly in water scarce areas. Decision-makers are encouraged to include the WFP of management practices in selecting among MSW management practices.

Finally, decisions makers are encouraged to assess the multiple criteria associated with selecting an alternative fuel. The overall analysis using the environmental and financial criteria showed that conventional-diesel, followed by hydraulic-hybrid WCVs are the best alternatives, followed by LFG-sourced natural gas, fossil natural gas, and biodiesel. Environmentally, hydraulic-hybrid and fossil natural gas, performed better than conventional-diesel; however, the vehicle cost of hydraulic-hybrid affected the financial ranking, although fuel price savings were observed. A policy that rewards decision-makers to buy more expensive vehicles is needed. The environmental benefits of hydraulic-hybrid exceeds diesel, however the initial cost of the vehicles ranked it behind. The elimination of the fueling station criterion from the financial decision level ranked landfill gas (LFG) sourced natural gas as the best option; suggesting that LFG-sourced natural gas is the best alternative to fuel WCV when accessible. The overall ranking of diesel and natural gas was found to be more sensitive to changing fuel price as compared to LFG-sourced natural gas, BD20 or BD100.

6.2. Future Research Recommendations

The social factors affecting low Florida recycling participation rates was not addressed in this study. Further research is needed to address social factors affecting recycling. Moreover, municipalities are encouraged to report recycling participation and set-out rates in Florida more frequently. A study that addresses the impact of pay-as-you-throw is also recommended.

The expansion of the WFP calculation to include innovative MSW management practices (e.g. MSW to ethanol or biodiesel), providing a new criteria to compare waste-based biofuels with agricultural-based and other fuels. This new assessment tool will be helpful to clearly identify the impact of fuels on water resources. Moreover, this study presented the WFP of a few recyclables commodities and should be expanded to include others.

APPENDIX: SUPPLEMENTARY TABLES

Table A.1: Reported tonnage of waste collected by Floridian RCC programs.

No	Program	Household Count (N_T)	Recycling System	Recycling Container	Group	Year	Garbage Collected W_G (Metric Ton ((MT)	Recyclables Collected W_R (MT)	Yard Waste Collected (MT)
1	1G,1R,1YW	8,155	DS	Bins†	B	2012	5,101*	1,133*	1,880*
2	2G,1R,1YW	17,000	DS	Bins†	A	2012	20,016	1,407	5,822
3	2G,1R,1YW	22,500	DS	Bins†	A	2012	18,694	4,490	7,259
4	2G,1R,1YW	4,200	DS	Bins†	A	2012	4,534	1,026	1,241
5	2G,1R,1YW	38,293	DS	Bins†	A	2012	29,394	4,746	6,513
6	1G,1R,1YW	69,812	SS	240 Liter (64 gallon) toter	D	2012	91,133	30,870	36,668
7	2G,1R,1YW	3,258	DS	Bins†	A	2012	4,226	233	909
8	2G,1R,1YW	4,700	DS	Bins†	A	2012	5,085	533	1,210
9	2G,1R,1YW	1,040	DS	Bins†	A	2012	1,447	221	64
10	2G,1R,1YW	11,434	DS	Bins†	A	2012	10,963	1,490	2,555
11	2G,1R,1YW	8,900	SS	340 Liter (90 gallon) toter	C	2012	8,980	1,551	2,875
12	2G,1R,1YW	12,900	DS	Bins†	A	2012	12,798	1,143	2,643
13	1G,1R,1YW	33,865	DS	Bins†	B	2012	27,901	3,386	5,770
14	2G,1R,1YW	7,400	DS	Bins†	A	2012	6,789	756	1,592
15	2G,1R,1YW	40,087	DS	Bins†	A	2012	39,115	3,418	9,545
16	2G,1R,1YW	35,924	DS	Bins†	A	2012	34,056	5,117	10,023
17	2G,1R,1YW	40,640	DS	Bins†	A	2012	34,940	4,166	8,595
18	2G,1R,1YW	40,402	DS	Bins†	A	2012	38,882	4,746	7,944
19	2G,1R,1YW	42,478	DS	Bins†	A	2012	33,693	5,742	7,550
20	2G,1R,1YW	10,589	DS	Bins†	A	Oct 09 - Sep10	9,330	2,166	3,347
21	1G,1R,1YW	10,784	DS	Bins†	B	Oct 10 - Sep11	9,806	2,393	3,504
22	1G,1R,1YW	4,500	SS	240-liter (64-gallon) toter	D	2011	3,112	1,650	667
23	2G,1R,1YW	1,400	SS	240-liter (64-gallon) toter	C	2011	968	513	207
24	2G,1R,1YW	2,100	SS	240-liter (64-gallon) toter	C	2011	1452	770	311
25	2G,1R,1YW	1,100	SS	240-liter (64-gallon) toter	C	2011	761	403	163

*Less than one year tonnage, therefore it was used only to evaluate recycling efficiency
†60-liter (16-gallon) bins
A: 2G, 1R, 1YW-DS; B: 1G, 1R, 1YW-DS; C: 2G, 1R, 1YW-SS; D: 1G, 1R, 1YW-SS

Table A.2: Composition of the recovered material from SS trucks and MRF output.

Material	Collection Truck Composition* (% of total weight)	MRF Output* (% of total weight)
Amber Glass	6.86	0.02
Clear Glass	8.63	0.03
Green Glass	4.11	0.02
HDPE Colored Containers (Baled)	2.25	0.89
HDPE Natural Containers (Baled)	1.53	0.73
LDPE Film (Baled)	N/A	0.33
Mixed Papers (Baled)	22.4	2.14
Mixed Rigid Plastic (Baled)	N/A	0.39
OCC (Baled)	10.7	0.24
OCC-BL_Baled		14.5
OCC (Baled)		13.5
PET Containers Comingled (Baled)	6.34	2.23
Plastic 1 Thru 7 (Baled)	2.12	0.35
Plastic 3 Thru 7 (Baled)		0.19
Polycarbonate	N/A	0.01
Polycarbonate (Del)	N/A	0.00
Polystyrene	N/A	0.03
Scrap Aluminum (loose)	1.14	0.00
Sorted Office Waste (Baled)	N/A	0.21
Special De Ink New #8 (Baled)	19.5	21.7
Special De Ink New #8 (Baled)		15.8
Steel Cans (Baled)	N/A	1.50
Three Mix Glass	N/A	15.5
Titanium	N/A	0.00
Used Beverage Cans (Baled)	N/A	0.70
Tin Cans	2.5	N/A
Residue	12.1	9.07

N/A: Not applicable
*Based on MRF operators input

Table A.3: Composition of the recovered material from DS trucks and MRF output.

Material	Collection Truck Composition* (% of total weight)	MRF Output* (% of total weight)
Aluminum	48	0.8
PET/HDPE		0.4
Mixed Plastic		12.3
Mixed Glass		21.3
Ferrous		2.2
Newspaper	52	10.2
Cardboard		9.7
Mixed Paper		32
Single Stream	N/A	0.5
Residue	N/A	10.4

N/A: Not Applicable,

*Based on MRF operators input

Table A.4: TSS, NH_3 and Heavy Metals Concentrations in Landfill Leachate for Traditional, Bioreactor, Ash Landfills (EREF, 1997; U.S. EPA, 1990; U.S. EPA, 2011b).

Leachate Constituent	Engineered and Bioreactor Landfills	Ash Landfill
	Concentration (mg/L)	
TSS	57	N/A
NH_3	343	12
Heavy Metals Concentration (µg/l)		
Arsenic	29-30	66.5-190
Cadmium	2.5-7	1.6-2.6
Chromium	52-85	12.5-20
Lead	5.7-13	13.8-28.3
Mercury	0.1-0.42	0
Selenium	2.5-9.7	50.2-160
Silver	12.5-66	0
Zinc	N/A	57.2-57.61
Copper	N/A	5.3-8.4
Iron	N/A	2700

N/A: Not Applicable,

Table A.5: WFP of the U.S. Electric power in 2013.

Energy mix in 2013 (U.S. EIA, 2014)		L per kWh generated by the fuel			L per kWh on the U.S. Grid	
		Lower Limit	Upper Limit	Reference	Lower Limit	Upper Limit
Coal	39.1%	0.54	2.09	Hill and Younos (2007)	0.21	0.82
Pet Coke	0.3%	15.49	31.05	Hill and Younos (2007)	0.05	0.10
Oil	0.3%	15.49	31.05	Hill and Younos (2007)	0.05	0.10
Natural Gas	27.4%	0.36	0.36	Gleick (1994)	0.10	0.10
Other Gas	0.3%	0.36	0.36	Gleick (1994)	0.00	0.00
Nuclear	19.4%	1.52	2.74	Jacobson (2009)	0.29	0.53
Hydro	6.5%	79.42	79.42	Gerbens-Leenes et al., 2009a	5.18	5.18
Other	0.3%	0.00	0.00	n/a	0.00	0.00
Wind	4.1%	0.00	0.00	Jacobson (2009)	0.00	0.00
Solar	0.2%	0.13	2.82	Jacobson (2009)	0.00	0.01
Biomass	1.5%	133.57	151.62	Gerbens-Leenes et al., 2009b	1.97	2.24
Geothermal	0.4%	0.02	0.02	Jacobson (2009)	0.00	0.00
Total	100%				7.86	9.08

Table A.6: GHG Emissions of the U.S. electric power in 2013.

Energy mix in 2013 (U.S. EIA , 2014)		Kg CO_{2eq} per kWh generated by the fuel			Kg CO_{2eq} per kWh on the U.S. Grid	
		Lower Limit	Upper Limit	Reference	Lower Limit	Upper Limit
Coal	39.1%	1	1	Kaplan et al. (2009)	0.391	0.391
Pet Coke	0.3%	1	1	Kaplan et al. (2009)	0.003	0.003
Oil	0.3%	0.89	0.89	Kaplan et al. (2009)	0.003	0.003
Natural Gas	27.4%	0.44	0.44	Kaplan et al. (2009)	0.121	0.121
Other Gas	0.3%	0.44	0.44	Kaplan et al. (2009)	0.001	0.001
Nuclear	19.4%	0.016	0.055	Kaplan et al. (2009)	0.003	0.011
Hydro	6.5%	0.0042	0.152	Kaplan et al. (2009)	0.000	0.010
Other	0.3%	--	--	n/a	0.000	0.000
Wind	4.1%	0.0046	0.0554	Fthenakis and kim (2007)	0.000	0.002
Solar	0.2%	0.022	0.049	Fthenakis and kim (2007)	0.000	0.000
Biomass	1.5%	0.035	0.037	Hartmann and Kaltschmitt (1999)	0.0005	0.0005
Geothermal	0.4%	0.081	0.081	Holm et al. (2012)	0.000	0.000
Total	100%				0.52	0.54

Table A.7: WFP and GHG of Recovered Energy (Data for Figure 4.5)

	WFP (L per kWh)	References	GHG Emissions (kg CO2eq per kWh)	References
US Electric Grid 2013	7.89-9.08	This Study	0.55	This Study
Traditional Landfills	4.5-13.5	This Study	2.3-5.5	Kaplan et al. (2009)
Bioreactor Landfills	1.6-3.0	This Study	2.3-5.5	Kaplan et al. (2009)
Waste Combustion	0.7	This Study	0.4-1.5	Kaplan et al. (2009)
Coal	0.54-2.09	Hill and Younos (2007)	1	Kaplan et al. (2009)
Natural Gas	0.36	Gleick (1994)	0.44	Kaplan et al. (2009)
Oil	15.5-31	Hill and Younos (2007)	0.89	Kaplan et al. (2009)
Nuclear	1.52-2.74	Jacobson (2009)	0.016-0.055	Kaplan et al. (2009)
Solar Energy	0.13-2.82	Jacobson (2009)	0.022-0.049	Fthenakis and kim (2007)
Wind	0.004	Jacobson (2009)	0.0046-0.0554	Fthenakis and kim (2007)
Geothermal	0.018	Jacobson (2009)	0.081	Holm et al. (2012)

Table A.8: WPF of Recycled Commodities

Commodity	Virgin-based Manufacturing WFP				Reference	Recycling losses (%)	Recycled Commodity WFP			
	m^3 per MT						m^3 per MT of recycled material			
	Blue WFP	Green WFP	Grey WFP	Total WFP			Blue WFP	Green WFP	Grey WFP	Total WFP
Paper	33	1980	1287	3300	UPM-Kymmene (2011)	40	-20	-1980	-1542	-68640
Tin Cans	4	0	61	65	Van der Leeden et al. (1990)	2	-4	0	-64	-65
Aluminum Cans	33	0	24	57	Li (2010)	7	-31	0	-32	-57
HDPE	10	0	227	237	Van der Leeden et al. (1990)	14	-9	0	-247	-237

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