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Environmental Sustainability of Wastewater Treatment Plants Integrated with Resource Recovery: The Impact of Context and Scale

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by

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of the requirements for the degree of
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DEDICATION

This dissertation is dedicated to my wife, Aviana Xochitl, and our newborn son, Calixto Ollin. I truly could not have done this without your love and support. This is also dedicated to my parents, my brothers, my grandparents, my extended family, my ancestors, the elders, the planet and our future generations.

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ABSTRACT

There is an urgent need for wastewater treatment plants (WWTPs) to adapt to a rise in water and energy demands, prolonged periods of drought, climate variability, and resource scarcity. The Environmental Protection Agency's (EPA) strategic research action plan states that the "failure to manage the Nation's waters in an integrated sustainable manner will limit economic prosperity and jeopardize human and aquatic ecosystem health" (EPA, 2012a). As population increases, minimizing the carbon and energy footprints of wastewater treatment, while properly managing nutrients is crucial to improving the sustainability WWTPs. Integrated resource recovery can mitigate the environmental impact of wastewater treatment systems; however, the mitigation potential depends on various factors such as treatment technology, resource recovery strategy, and system size.

Amidst these challenges, this research seeks to investigate the environmental sustainability of wastewater treatment plants (WWTPs) integrating resource recovery (e.g., water reuse, energy recovery and nutrient recycling) in different contexts (developing versus developed world) and at different scales (household, community, and city). The over-arching hypothesis guiding this research is that: Context and scale impact the environmental sustainability of WWTPs integrated with resource recovery. Three major research tasks were designed to contribute to a greater understanding of the environmental sustainability of resource recovery integrated with wastewater treatment systems. They include a framework development task (Chapter 2), scale assessment task (Chapter 3), and context assessment task (Chapter 4).

The framework development task includes a critical review of literature and models used to design a framework to assess the environmental sustainability of wastewater treatment and integrated resource recovery strategies. Most studies used life cycle assessment (LCA) to assess these systems. LCA is a quantitative tool, which estimates the environmental impact of a system over its lifetime (EPA, 2006). Based on this review, a comprehensive system boundary was selected to assess the life cycle impacts of collection, treatment, and distribution over the construction and operation and maintenance life stages. Additionally, resource recovery offsets associated with water reuse, energy recovery, and nutrient recycling are considered. The framework's life cycle inventory includes material production and delivery, equipment operation, energy production, sludge disposal, direct greenhouse gas (GHG) emissions, and nutrients discharged to the environment. Process-based LCA is used to evaluate major environmental impact categories, including global impacts (e.g., carbon footprint, embodied energy) and local impacts (e.g., eutrophication potential). This is followed by an interpretation of results using sensitivity or uncertainty analysis.

The scale assessment task investigates how scale impacts the environmental sustainability of three wastewater treatment systems integrated with resource recovery in a U.S. context. Household, community, and city scale systems using mechanized technologies applicable to a developed world setting were investigated. The household system was found to have the highest environmental impacts due high electricity usage for treatment and distribution, methane emissions from the septic tank, and high nutrient discharges. Consequently, the life cycle impacts of passive nutrient reduction systems with low energy usage at the household level merit further investigation. The community scale system highlights trade-offs between global impacts (e.g., embodied energy and carbon footprint) and local impacts (e.g., eutrophication potential) where low nutrient pollution can be achieved at the cost of a high embodied energy and carbon footprint. The city scale system had the lowest global impacts due to economies of scale and the benefits of integrating all three forms of resource recovery: Energy recovery, water reuse, and nutrient recycling. Integrating these three strategies at the city scale led to a 49% energy offset, which mitigates the carbon footprint associated with water reuse.

The context assessment task investigates how context impacts the environmental sustainability of selected community scale systems in both Bolivia and the United States. In this task, rural developing world and urban developed world wastewater management solutions with resource recovery strategies are compared. Less mechanized treatment technologies used in rural Bolivia were found to have a lower carbon footprint and embodied energy than highly mechanized technologies used in urban United States. However, the U.S. community system had a lower eutrophication potential than the Bolivia systems, highlighting trade-offs between global and local impacts. Furthermore, collection and direct methane emissions had more important energy and carbon implications in Bolivia, whereas treatment electricity was dominant for the U.S. community system. Water reuse offsets of embodied energy and carbon footprint were higher for the U.S community system, because high quality potable water is replaced instead of river water. In contrast, water reuse offsets of eutrophication potential were high for the Bolivia systems, highlighting the importance of matching treatment level to end-use application. One of the Bolivia systems benefits from the integration of water, energy, and nutrient recovery leading to beneficial offsets of both global and local impacts. This research can potentially lead to transformative thinking on the appropriate scale of WWTPs with integrated resource recovery, while highlighting that context lead to changes in the dominant contributors to environmental impact, appropriate technologies, and mitigation strategies.

CHAPTER 1: INTRODUCTION

1.1 Background

Global stressors, such as population growth, climate change, increasing urbanization, excessive nutrient inputs into surface waters, and water stress place additional pressure on water and wastewater utilities to provide adequate water and sanitation in an energy efficient manner, while protecting human health and the environment (Zimmerman et al., 2008). By 2050, the global population is expected to increase by 32% to 9.1 billion people (Evans, 2011). Increased population and affluence can coincide with a rise in water demand, which is estimated to increase electricity used to supply and treat water and wastewater by 33% by 2022 (ASE, 2002). Meanwhile, up to 23% of the total energy used within a typical municipality comes from wastewater treatment in some regions (CEC, 1992; Means, 2004). Additional materials and energy required to treat wastewater to higher standards while meeting increased demands contribute to larger environmental footprints and economic costs over the life cycle.

Water reuse and other forms of resource recovery (e.g., energy recovery and nutrient recycling) can help reduce the environmental impact associated with wastewater treatment facilities. Urban water demand, water scarcity, efficient resource utilization, and the protection of human and ecosystem health are additional drivers towards recent movements to reclaim water and other resources (EPA, 2012b; NRC, 2012). All of these drivers have led to the implementation of 3,300 water reclamation systems globally (FAO, 2010). From a systems perspective, water reuse can offset energy and resources needed for conventional water production, energy recovery can lead to energy offsets by replacing natural gas, and nutrient recycling can offsets chemical fertilizer usage (Fine and Hadas, 2012; Mihelcic et al., 2011; Mo and Zhang, 2012a). An estimated 22% of the world's phosphorus supply could be meet through nutrient recycling from urine and feces, which also leads to the reduction of anthropogenic impacts of phosphate mining, while addressing phosphorus scarcity (Mihelcic et al., 2011). Collectively, integrated resource recovery via water reuse, energy recovery, and nutrient recycling (see Figure 1) can address the challenges associated with the rising environmental footprint of wastewater treatment.

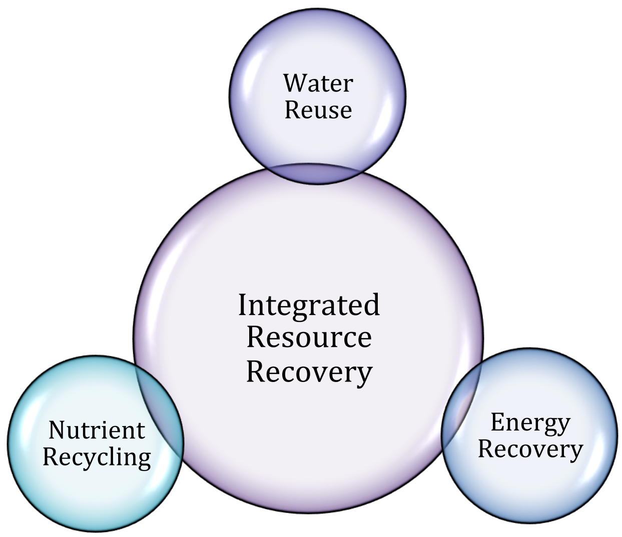

Figure 1. Diagram of integrated resource recovery including water reuse, nutrient recycling, and energy recovery

Many studies use life cycle assessment (LCA) to evaluate the carbon footprint and/or embodied energy of wastewater treatment plants (WWTPs) (Hospido et al., 2004), WWTPs with water reuse, nutrient recycling and/or energy recovery applications (Lundie et al., 2004; Meneses et al., 2010; Ortiz et al., 2007; Pasqualino et al., 2010; Tangsubkul et al., 2005; Tillman et al., 1998; Zhang et al., 2010; Cornejo et al., 2013; Cornejo et al., 2014) and water supply systems (e.g., comparing water reuse, desalination and importation) (Lyons et al., 2009; Stokes and Horvath, 2006, 2009; Santana et al., 2014). Eutrophication potential is also a frequently investigated environmental impact category, pertinent to the life cycle impacts of water reuse and wastewater systems (Dennison et al., 1998; Hospido et al., 2004; Lundie et al., 2004; Meneses et al., 2010; Muñoz et al., 2010; Pasqualino et al., 2010; Tangsubkul et al., 2005). Consequently, embodied energy, carbon footprint, and eutrophication potential were identified as key environmental sustainability impact categories related to the water-energy-carbon-nutrient nexus of wastewater management solutions and resource recovery strategies, as defined in Table 1.

Table 1. Description of embodied energy, carbon footprint, and eutrophication potential and key contributors to these environmental impact categories

Impact Category	**Description**	**Contributors**		
Embodied Energy	Life cycle energy consumption	N/A	Direct Energy (e.g., on-site energy)	Indirect energy (e.g., production of materials)
Carbon Footprint	Life cycle greenhouse gas emissions (GHG)	Direct GHG emissions (e.g., CH_4 and N_2O)	Indirect GHG emissions (e.g., electricity)	Other indirect emissions (e.g., production of materials)
Eutrophication Potential	Life cycle nutrient pollution	Direct sources (e.g., nutrients) discharged to environment	Indirect sources (e.g., NO_x from electricity)	Other indirect sources (e.g., production of materials)

Life cycle assessment (LCA) is a quantitative tool that estimates the environmental impact of a process or product over its life, including raw material extraction, construction, operation, reuse and end-of-life phases (EPA, 2006). Embodied energy is the life cycle energy consumption consisting of direct energy (e.g., on-site energy consumption from electricity and diesel) and indirect energy (e.g., infrastructure, chemicals). Carbon footprint represents the life cycle greenhouse gas (GHG) emissions consisting of: direct (Scope 1) emissions (e.g., CH_4 and N_2O), indirect (Scope 2) emissions (e.g., electricity production), and other indirect (Scope 3) emissions (e.g., infrastructure, chemicals). Eutrophication potential is the life cycle nutrient pollution that increases the risk of algal growth in water bodies impairing water quality, depleting oxygen levels, and impacting freshwater availability. Eutrophication comes from direct sources (e.g., nutrients discharged directly to the environment), indirect sources (e.g., NO_x from electricity) and other indirect sources (e.g., infrastructure, chemicals).

A brief overview of embodied energy and carbon footprint ranges from representative studies is shown in Table 2. Embodied energy and carbon footprint values from developed and developing world studies, as well as highly mechanized and less mechanized technologies integrating natural treatment processes are shown. Most of these studies took place in the developed world on mechanized wastewater treatment technologies. A general trend can be observed in this table on the high end of the ranges in which the embodied energy and carbon

Table 2. Review of embodied energy and carbon footprint ranges for representative LCA literature in developing and developed world regions and for highly mechanized and less mechanized wastewater treatment technologies with resource recovery

Context	**Countries**	**Embodied Energy Range (MJ/m^3)**	**Literature**	**Carbon Footprint Range (kg CO_2eq/m^3)**	**Literature**
Developed World	Australia, United States, Spain	1.3-23	Lundie et al. (2004); Stokes and Horvath (2006); Stokes and Horvath (2009); Meneses et al. (2010); Pasqualino et al. (2010)	0.12-1.8	Tangsubkul et al. (2005); Stokes and Horvath (2006); Lyons et al. (2009); Stokes and Horvath (2009); Meneses et al. (2010); Pasqualino et al. (2010)
Developing World	China, South Africa, Bolivia	3.6-4.7	Zhang et al. (2010); Cornejo et al. (2013)	0.33-0.63	Friedrich et al. (2009); Cornejo et al. (2013)
Highly Mechanized	Australia, United States, Spain	1.3-23	Lundie et al. (2004); Stokes and Horvath (2006); Stokes and Horvath (2009); Meneses et al. (2010); Pasqualino et al. (2010); Zhang et al. (2010)	0.12-1.8	Stokes and Horvath (2006); Lyons et al. (2009); Stokes and Horvath (2009); Meneses et al. (2010); Pasqualino et al. (2010)
Less Mechanized	Australia, Bolivia	3.6-4.1	Cornejo et al. (2013)	0.33-0.63	Tangsubkul et al. (2005); Cornejo et al. (2013)

footprint of developed world technologies are higher than developing world technologies. Additionally, the embodied energy and carbon footprints of highly mechanized technologies on the high end of the ranges are higher than less mechanized technologies integrating natural treatment processes.

Whereas a wide range of previous studies have documented embodied energy and carbon footprint, fewer studies have documented eutrophication potential of WWTPs with integrated resource recovery. Variations in methodology and presentation of results limit adequate comparisons from previous LCA literature for eutrophication potential, though a general range of 0.03 g PO_4eq/m^3 to 1.00 g PO_4eq/m^3 was identified (Meneses et al., 2010; Pasqualino et al., 2010). The range emerges from indirect sources of eutrophication only (e.g. NO_x from electricity), where the low end of the range is from an agricultural reuse scenario (tertiary treatment with fertilizer offsets) and the high end of the range is from a potable water reuse scenario (WWTPs with tertiary treatment), which considers a more comprehensive system boundary.

In spite of these general trends, the comparison of life cycle assessment results from different studies is difficult because inconsistent LCA frameworks are implemented for analysis. Variations in system boundaries, phases considered, parameters considered, technologies evaluated, underlying assumptions, electricity mixes, and estimation methodologies lead to a wide range of findings from different literature sources. Consequently, a consistent framework is required to better compare resource recovery technologies at different scales of implementation (e.g., different levels of centralization) in different contexts (developed versus developing world). In this research scale refers to the size of a system or level of centralization (e.g., household, community, city scale systems), whereas context refers to location and factors specific to a given region that impact wastewater management (i.e., socio-political conditions, regulations, decision-making processes, economics, demographics, operational capacity, social acceptance, appropriate treatment technologies selection, resource recovery strategies implemented, etc.).

Limited research has investigated how scale of implementation or level of centralization impacts the environmental sustainability of WWTPs that are integrated with resource recovery. Most of the previous studies on the impact of scale have focused on scale's influence on system cost. Cost studies have shown that wastewater treatment systems adhere to cost-based economies of scale, in which centralized systems provide cost saving compared to decentralized systems (EPA, 1978a, 1978b; Fraquelli and Giandrone, 2003; Hopkins et al., 2004). Concerns over rising energy costs, climate change, and the protection of local water bodies; however, have led to an increase in research on scale's impact on the environmental footprint of WWTPs integrated with resource recovery.

Life cycle assessment studies investigating the impact of scale on the environmental sustainability of wastewater treatment systems have focused on hypothetical source separation schemes and sludge management options in a European context (Dennison et al., 1998; Tillman et al., 1998; Lundin et al., 2000), as shown in Table 3. European studies have found that source separation schemes and sludge management adhere to environmentally-based economies of scale, where centralization is beneficial to reducing the environmental impact. However, some limitations in these European studies are the exclusion of direct emissions (e.g., methane and nitrous oxide) and/or the exclusion of comprehensive life cycle assessment (i.e., only conducting a life cycle inventory).

Table 3. Summary of key studies assessing scale's impact on WWTPs with resource recovery applications

Source	Location	Description	Findings	Research Gap
Tillman et al. (1998)	Sweden	LCA on two WWTPs with two decentralization alternatives (filter bed and urine separation)	Increased decentralization decreased electricity usage, but increased fossil fuel usage	Excludes direct emissions (e.g., methane and nitrous oxide)
Dennison et al. (1998)	United Kingdom	LCA on sludge management options for fifteen wastewater treatment facilities	Complete centralization reduced the carbon footprint of sludge handling	LCI only. Excludes infrastructure
Lundin et al. (2000)	Sweden	LCA comparing two WWTPs with two separation schemes (liquid composting and urine separation)	Source separation adheres to environmentally-based economies of scale	LCI only. Focuses on source separation. Excludes water reuse
Pitterle (2009)	United States	LCA on six WWTPs ranging from 100 gpd to 130 mgd in Colorado	Benefits to centralization due to economies of scale	Doesn't fully assess integrated resource recovery
Shehabi et al. (2012)	United States	LCA on two WWTPs in California	Benefits to centralization due to economies of scale	Doesn't assess scale's influence on eutrophication potential

Similarly two U.S. based studies found environmental benefits of centralization in wastewater management (Pitterle, 2009; Shehabi et al., 2012). The U.S. studies address global concerns (e.g., carbon footprint, embodied energy), but ignore local concerns (e.g., eutrophication potential of local water bodies). Furthermore, most U.S. and European studies don't fully assess integrated resource recovery alternatives. For example, most studies exclude water reuse or fail to consider nutrient recycling from reclaimed water. Consequently, further research is needed on the environmental impacts of integrated water, energy and nutrient recovery at different scales using a comprehensive framework that considers global (e.g., carbon footprint and embodied energy) and local concerns (e.g., eutrophication potential of local water bodies).

Furthermore, few studies focus on the life cycle environmental impact of wastewater treatment systems with resource recovery in a developing world context or comparisons between systems in developing and developed world settings. Galvin (2013) investigated the life cycle impacts of household wastewater management systems with nutrient recycling and energy recovery, but excludes water reuse. This study highlights the benefits of energy recovery from on-site biogas digesters and fertilizer offsets, which effectively achieve carbon neutrality; however, another study found that on-site biogas recovery has a high failure rate in the developing world leading to unintended methane releases caused by improper operation and maintenance practices (Bruun et al., 2014). Other LCA studies in the developing world focus on household water provision in Mali, West Africa (Held, 2013), shea butter production in Ghana, West Africa (Adams, 2015) and large-scale mechanized water reclamation facilities (greater than 10 mgd) serving urban areas in China (Zhang et al., 2010) and South Africa (Friedrich et al., 2009).

For smaller-scale household or community scale applications (<5 mgd); however, Muga and Mihelcic (2008) suggest that mechanized treatment technologies (e.g., activated sludge processes) are less appropriate than natural systems (e.g., waste stabilization ponds (WSPs)) in the developing world, due to higher costs and higher energy-intensities. Furthermore, Verbyla et al. (2013), highlights the benefits of water reuse and nutrient recycling for food security from community scale waste stabilization ponds in rural Bolivia. Other life cycle assessment studies on the carbon footprint of WSPs have been conducted in urban areas such as Sydney, Australia (Tangsubkul et al., 2005). However, limited research has been conducted on both global (e.g., embodied energy, carbon footprint) and local (e.g., eutrophication potential) life cycle environmental impacts of community-managed wastewater systems integrated with resource recovery in rural developing regions. Additionally, to the author's knowledge no peer-reviewed studies assessed the impact of context (e.g., developed versus developing world) on the environmental sustainability of community-scale wastewater management systems integrated with resource recovery.

1.2 Scope of Research

Consequently, this research seeks to investigate the influence that context (e.g., rural developing world setting versus urban developed world setting) and scale (e.g., size of system or level of centralization) have on the environmental sustainability of appropriate wastewater treatment technologies that recover water, energy, and nutrient resources. The central hypothesis guiding this research is that: Context and scale impact the environmental sustainability of integrated resource recovery systems applied to management of wastewater. A framework was developed to identify proper models and methods to investigate systems in both developed and developing world settings. Then, life cycle assessment (LCA) case studies were conducted to test the stated hypothesis. Context is expected to impact the environmental sustainability of wastewater treatment technologies and resource recovery strategies because location leads to changes in appropriate technologies for a given region and rural developing communities manage wastewater systems differently than urban developed regions. Therefore context related factors such as location, socio-political conditions, operational requirements, technology implemented, resource recovery strategies, and other demographics are expected to change, impacting the environmental sustainability of varying systems. Similarly, scale of implementation is expected to impact the environmental sustainability of these systems. Environmentally-based economies of scale, as well as changes in wastewater treatment technologies and resource recovery strategies applicable at each scale are expected to lead to changes in environmental sustainability at varying scales.

The contributions of construction (e.g., production of materials) and operation phases (e.g., direct emissions, electricity usage) are expected to be context and scale dependent. Previous research has highlighted some differences in construction and operation phases for mechanized systems versus less mechanized systems integrating natural treatment processes (Cornejo et al., 2013) and systems implemented at different scales (Pitterle, 2009). These studies found that the environmental impact of infrastructure had a higher relative contribution for systems with natural treatment technologies and smaller systems since less electricity is typically used for these systems. In contrast, operation and maintenance had a higher environmental impact for mechanized systems at larger scales, due higher levels of electricity usage. Furthermore, scale and context are expected to lead to different resource recovery strategies that alter the offset or mitigation potential of environmental impact categories. A summary of research conducted for this dissertation is shown in Figure 2 and a diagram of the research conducted in this dissertation is shown in Figure 3.

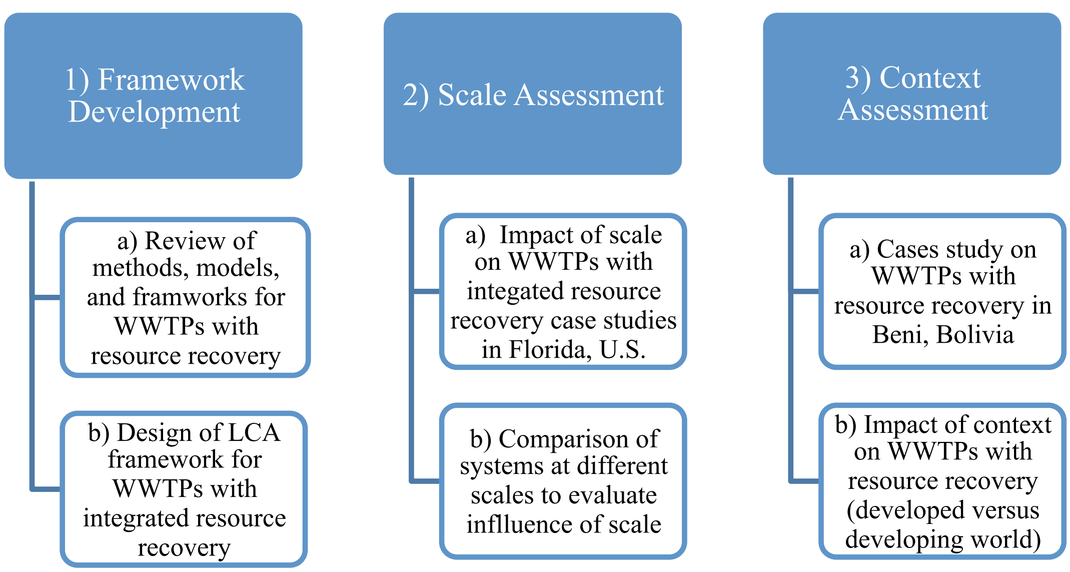

Figure 2. Major research tasks including framework development task used to design an LCA framework for wastewater treatment plants with integrated resource recovery, scale assessment task used to evaluate the impact of scale in United States and context assessment task used to compare systems in Bolivia and United States

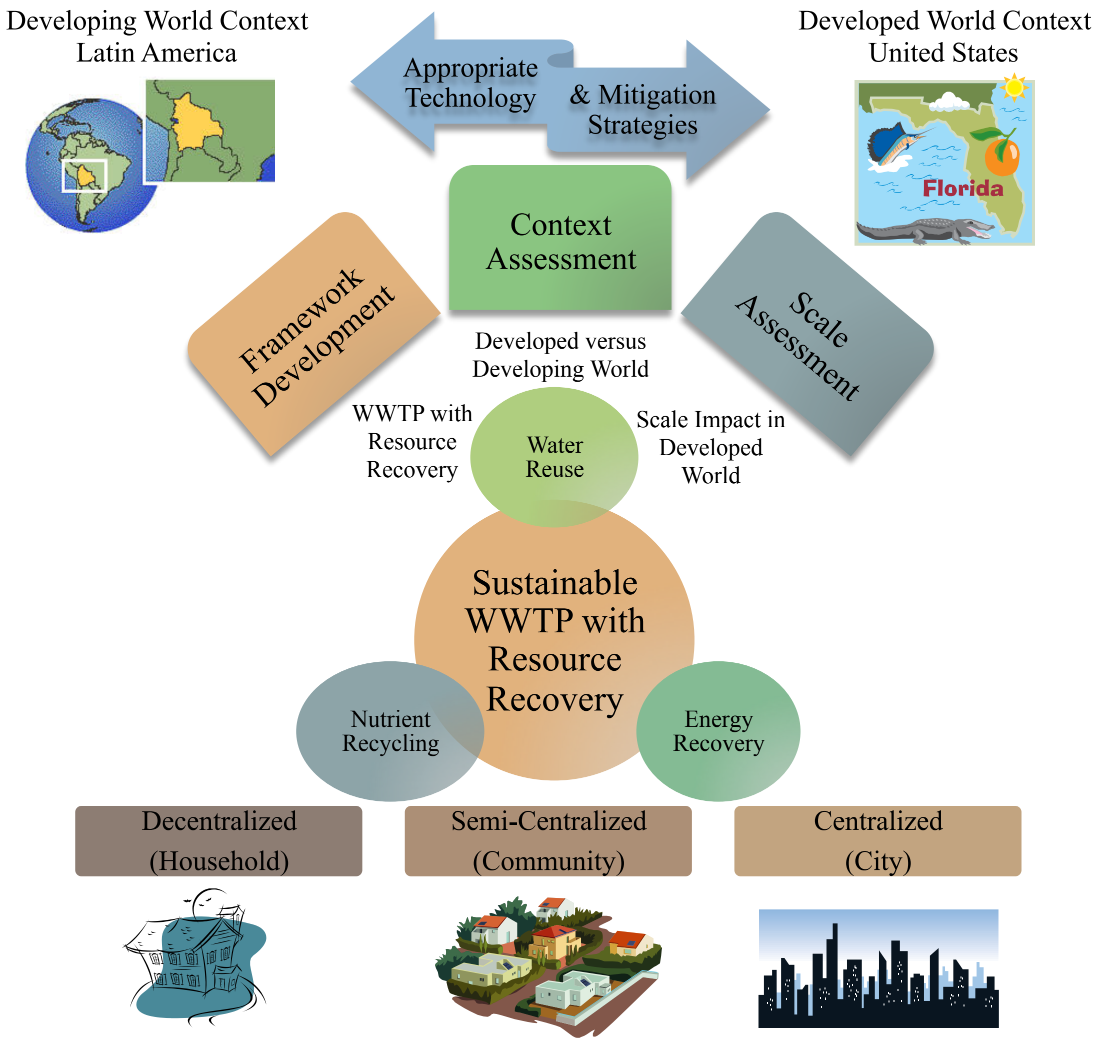

Figure 3. Scope of research investigating the impact of context and scale on the environmental sustainability of wastewater treatment systems integrated with resource recovery

This research consists of three major research tasks and a concluding chapter to summarize key findings. The following tasks are conducted to test the stated hypothesis:

- Framework Development (Chapter 2): Develop a life cycle assessment (LCA) framework that is appropriate for wastewater treatment plants (WWTPs) that are integrated with resource recovery (water reuse, energy recovery, and nutrient recycling).
- Scale Assessment (Chapter 3): Assess the impact of scale on the environmental sustainability of resource recovery systems integrated with wastewater treatment at a household, community, and city scale in Florida, United States.

- Context Assessment (Chapter 4): Assess the impact of context on the environmental sustainability of wastewater treatment integrated with resource recovery systems by comparing community scale systems in Bolivia and United States.

1.3 Summary of Technology Selection

Three systems are assessed in a developed world context and two systems are assessed in a developing world setting. A summary of implementation scale selected for technologies in different settings is summarized in Table 4. Developed world technologies selected focus on a large urban setting in a coastal region, whereas developing world technologies focus on a small town near rural agricultural areas. These regions represent critical areas for research on the water-energy-nutrient nexus, as they are expected to face population growth and increases in water demand with increased urbanization (Caplan and Harvey, 2010; Hallegatte et al., 2013).

Selection criteria for U.S and Bolivia systems include: (1) Data availability, and (2) Commonly-used and proven resource recovery applications. Additionally, U.S. systems are applicable and appropriate to an urban developed world context, whereas Bolivia systems are appropriate and applicable to rural developing world context.

Table 4. Level of centralization, scale, population equivalent, and flowrate for technologies evaluated in this research. Includes household, community, and city scale systems in United States and community scale systems in Bolivia

Level	Scale	Population equivalents (p.e.)	Flowrate (mgd)	Developing World Technologies	Developed World Technologies
Decentralized	Household	2-3	Less than 0.02		X
Semi-centralized	Community	~1,000	0.02-0.3	X	X
Centralized	City	100,000	10.3		X

Note: X indicates technologies will be assessed at given scale

A summary of the technologies analyzed in this research is provided in Table 5. Developed world technologies selected in Florida, U.S. include: (1) a 250 gallon per day (gpd) household septic tank followed by an aerobic treatment unit, and drip irrigation for reuse, (2) a 0.31 million gallons per day (mgd) community water reclamation facility with nitrification/denitrification using headworks (grit removal, bar screens, odor scrubbing), equalization tanks, aeration tanks, denitrification tanks, re-aeration, clarifiers, denitrification filters, a clearwell, chlorination and UV disinfection, aerobic digestion, and landscape irrigation for reuse, (3) a 10.3 mgd city scale advanced water reclamation facility with headworks (grit removal, bar screens), activated sludge (biological secondary treatment includes aeration basins with return activated sludge for biochemical oxygen demand (BOD) removal), secondary clarification, filtration, chlorination, anaerobic digestion for energy recovery, and landscape

Table 5. Summary of technologies analyzed

Context	Scale	Population served	Treatment Processes	Resource Recovery Strategies
Developed World Systems	Household (250 gpd)	2-3	Primary tank, secondary (aerobic treatment unit)	Water and nutrient reuse for subsurface landscape drip irrigation, nutrient recycling from biosolids
	Community (0.31 mgd)	~1,500	Headworks, aeration tanks, denitrification tanks, re-aeration, clarification, de-nitrification filters, a clearwell, chlorination, UV, aerobic digestion	Water and nutrient reuse for golf course irrigation, nutrient recycling from biosolids
	City (10.3 mgd)	100,000	Headworks, activated sludge, (aeration with return activated sludge), secondary clarification, filtration, chlorination, anaerobic digestion	Water and nutrient reuse for landscape irrigation, nutrient recycling from biosolids, energy recovery
Developing World Systems	Community (0.019 mgd)	~1,471	Grit removal chamber, upflow anaerobic sludge blanket (UASB) reactor, two maturation ponds	Water and nutrient reuse for crop irrigation, energy recovery
	Community (0.024)	~727	Facultative Pond, two maturation ponds	Water and nutrient reuse for crop irrigation

irrigation for reuse. Developing world technologies selected in Beni, Bolivia include: (1) a 0.019 mgd UASB-Pond system (Upflow anaerobic sludge blanket reactor followed by two maturation lagoons in series) and (2) a 0.024 mgd 3-Pond system (A facultative pond followed by two maturation ponds in series) at the community scale. These technologies enable a comparison across both scale (system size) and context (technology).

In this research, commonly used resource recovery strategies applicable to small towns in Latin America and residential urban communities in the United States are selected for analysis. Integrated resource recovery includes water reuse, nutrient recycling, and energy recovery, where applicable at each scale. Water reuse and nutrient recycling via biosolids and/or reclaimed water are feasible or currently practiced at all the systems investigated. Energy recovery is feasible at the city scale U.S. based system and the community scale UASB-Pond system in Bolivia.

1.4 Framework Development Summary

The framework development stage (Chapter 2) consists of a thorough review of frameworks, methods, and models to assess the environmental impact of wastewater treatment and resource recovery strategies. The central purpose of this task is to develop a comprehensive LCA framework for WWTPs with integrated resource recovery systems. After synthesizing data on system boundaries, phases considered, input data requirements, emission sources considered, major environmental impact categories relevant to resource recovery, and appropriate assessment methods, an LCA framework for resource recovery applications is proposed. The proposed framework is used to assess the impact of scale (Chapter 3) and context (Chapter 4). This task has the dual purpose of gaining an in-depth understanding of the proper framework used to assess WWTPs and resource recovery systems in general, while developing the specific framework and assessment methodology utilized in the subsequent chapters.

A critical review of models and methods provides a thorough assessment of aspects needed to develop a comprehensive, robust, and transferable framework. Chapter 2 provides a detailed analysis of existing framework system boundaries, data sources, model inputs, methods for calculation, model outputs, limitations and applicability to WWTPs with integrated resource recovery. Outcomes of this research include: (1) a literature review of existing LCA and non-LCA frameworks related to wastewater treatment systems and resource recovery strategies and (2) a proposed framework for future research. This task addresses the following research questions:

- What should be included in the system boundary and what phases should be considered for wastewater treatment and resource recovery systems?
- What input data and emission sources should be considered for these systems?
- What are the main environmental impact categories associated with these systems?
- What should be included in an LCA framework that can assure consistency and robustness?
- What methods should be used to assess the offset potential of resource recovery?
- What are the major impacting factors of these systems?
- Are certain methods more appropriate to use in certain contexts (developing versus developed world)?

1.5 Scale Assessment in the Developed World

Chapter 3 assesses how scale influences the environmental sustainability of wastewater treatment systems implementing resource recovery in the developed world. The environmental impact of wastewater treatment integrated with resource recovery alternatives are evaluated at varying scales in the Tampa Bay region of Florida, a coastal urban area facing growing population and urbanization (Hallegatte et al., 2013). Specifically the carbon footprint, embodied energy and eutrophication potential of case studies at decentralized (household level), semi-centralized (community level), and centralized (city level) scales are assessed. The environmental sustainability of these systems, offset potential of resource recovery strategies, and trends associated with scale changes are evaluated in this chapter. The central hypothesis guiding this research task is that scale impacts the environmental sustainability of wastewater treatment systems and resource recovery strategies. The following research questions are addressed by conducting this research:

- How does scale impact technology selection and resource recovery solutions in a developed world settings?
- How does scale impact the environmental sustainability of resource recovery for major impact categories selected (e.g., carbon footprint, embodied energy, and eutrophication potential)?
    - How does scale lead to embodied energy differences between direct and indirect energy (or construction and operation phase)?
    - How does scale lead to carbon footprint differences between direct and indirect emissions (or construction and operation phase)?
    - How does scale impact eutrophication differences between direct and indirect sources of eutrophication potential?
- How do resource recovery strategies mitigate the impact wastewater treatment management at different scales?

1.6 Context Assessment: Developed versus Developing World Settings

Chapter 4 assesses the impact of context through a comparative analysis of the environmental sustainability of resource recovery technologies in both developed and developing world settings. The context assessment task (Chapter 4) includes two case studies of community scale WWTPs in rural Bolivia. These case studies are subsequently compared to the community scale wastewater treatment system with resource recovery in U.S. assessed in Chapter 3 allowing for a comparison of systems from both developing and developed world settings. The central hypothesis guiding this research task is that context impacts the environmental sustainability of wastewater treatment systems and resource recovery strategies. This chapter addresses the following research questions:

- How does context impact technology selection and resource recovery in developed and developing world settings?
- How does context impact the environmental sustainability of resource recovery for major impact categories selected (e.g., carbon footprint, embodied energy, and eutrophication potential)?
    - How does context lead to embodied energy differences between direct and indirect energy (or construction and operation phase)?
    - How does context lead to carbon footprint differences between direct and indirect emissions?
    - How does context impact eutrophication between direct and indirect sources of eutrophication potential?
- How does context impact the environmental sustainability of resource recovery?
- What knowledge can be transferred to improve sustainability of systems in both settings?

1.7 Significance

The Environmental Protection Agency's (EPA) Safe and Sustainable Water Resources strategic action plan states that research is needed on, "the minimization of energy use, effective recycling and re-use of water and waste, with the ultimate goal of providing communities with management options for sustainable water quality and availability" (EPA, 2012a). This investigation addresses these issues, aiming to provide insight to engineers and decision-makers on appropriate scale and/or design of the recovery of resources from wastewater in different settings. By focusing on developed and developing world settings, this project is also consistent with the EPA's mission to ensure, "the United States plays a leadership role in working with other nations to protect the global environment" (EPA, 2014a). The research applies an operational model for sustainable development that uses global partnerships, enhanced by integrating the best and most appropriate knowledge, methodologies, techniques, principles, and practices from both the developed and developing worlds (Mihelcic et al., 2007). Outputs from this research are based on sound science and provide practical quantification of the preferred outcomes of recovery and reuse that achieve social, economic, and ecological well-being associated with more sustainable wastewater management for current and future generations. This research provides insight on the how wastewater management solutions with resource recovery strategies can be applied at different scales and in different contexts to achieve environmentally sustainable solutions.

1.8 Broader Impacts

As seen in Table 6, research on WWTPs with integrated resource recovery encompass several key grand challenges for engineering put forth by the National Academy of Engineering (NAE). Resource recovery strategies that address NAE grand challenges include: (1) energy

Table 6. Resource recovery strategies that address engineering grand challenges

National Academy of Engineers Grand Challenges (NAE, 2012)	**Example of resource recovery strategy that address Grand Challenges**
Providing affordable and renewable energy	Energy recovery from anaerobic processes
Managing the nitrogen cycle	Nutrient recycling and reduced fertilizer use
Providing clean water	Potable water replacement via water reuse

recovery from anaerobic processes providing affordable and renewable energy sources (2) nutrient recycling and reduced fertilizer use leading to improved management of the nitrogen cycle and (3) water reuse replacing potable water leading to the provision of clean water (NAE, 2012). Research in the developing world also addresses key millennium development goals, such as, ensuring environmental sustainability (e.g., sanitation provision and reductions in global CO_2 emissions), reducing child mortality (e.g., addressing water quality issues) and enhancing global partnerships for development (UN, 2011). Additionally, this context-sensitive research on synergistic water-energy-nutrient systems can impact the current paradigm of wastewater management by transforming our understanding of wastewater as a resource, not a waste (Guest et al., 2009).

CHAPTER 2: FRAMEWORK DEVELOPMENT

2.1 Introduction

Stressors such as population growth, increased water demand, resource scarcity, and the impacts of climate change have led to a growing need for demand management and alternative water supplies, such as water reuse and desalination, in addition to innovative ways of recovering energy and nutrient resources. Worldwide, policy makers are increasingly adapting to climate variability and associated supply reliability issues (Major et al., 2011) because many parts of the world face periods of prolonged drought, population growth, and urbanization (Zimmerman et al., 2008; Padowski and Jawitz, 2012). For example, California recently issued the first mandatory water restriction in the state's history to address a four-year water crisis, in which drought conditions have drastically impacted the state's water resources (Nagourney, 2015). Wastewater treatment plants (WWTPs) with integrated resource recovery can provide a viable solution to address stressors on traditional water resources (e.g., groundwater and surface water supplies). Consequently, this chapter[1] provides a critical review of literature and frameworks on the environmental sustainability of WWTPs with integrated resource recovery (e.g., water reuse, energy recovery and nutrient recycling) to propose a comprehensive framework used for this dissertation. Integrated resource recovery has become more common worldwide to meet

[1]Portions of this chapter are adapted from *Journal of Water Reuse and Desalination* volume *04*, issue number *4*, pages *238-252*, with permission from the copyright holders, IWA Publishing. Additionally portions of this chapter are adapted from "Feasibility Study on Model Development to Estimate and Minimize Greenhouse Gas Concentrations and Carbon Footprint of Water Reuse and Desalination Facilities (WRRF-10-12)" and permission to reprint from the WateReuse Research Foundation was granted.

growing water demands, address resource scarcity, and move towards resiliency in water management.

Alternative water supplies are beneficial to water augmentation. Water reuse systems in particular are beneficial because they have the added value of incorporating other forms of resource recovery (e.g., energy recovery and nutrient recycling). Increased awareness and technological advancements have led to the implementation of 3,300 water reclamation systems globally (FAO, 2010), where water reuse has the potential benefit of protecting local water bodies from the risk of nutrient pollution. Although alternative water supplies increase water availability, in some cases they are more energy intensive than conventional water supply and treatment, due to higher levels of treatment and additional infrastructure needs. This raises concerns about the carbon footprint, embodied energy, and overall environmental sustainability of alternative water supplies. For instance, the embodied energy of drinking water provision in Tampa, Florida was estimated to be 7.2 megajoules per cubic meter of water treated (MJ/m^3) (Santana et al., 2014), whereas the embodied energy of water reuse and seawater reverse osmosis (RO) desalination were approximately 13-18 MJ/m^3 and 24-42 MJ/m^3, respectively (Lyons et al., 2009; Stokes and Horvath, 2009; Pasqualino et al., 2010). Carbon footprint values follow a similar trend, as desalination of seawater using RO (0.4-6.7 kg CO_2eq/m^3) is generally larger than water reuse (0.1-2.4 kg CO_2eq/m^3) (Cornejo et al., 2014).

Local concerns, such as the protection of water bodies and global concerns, such as carbon footprint's contribution to greenhouse gas (GHG) emissions are both important issues related to wastewater management. Worldwide, many local and state governments have taken action to mandate a reduction in GHG emissions to address the problem of elevated carbon footprints and climate change impacts. For example, since 2009 more than 825 cities are participating in the United States Mayors Climate Protection Agreement, which would reduce GHG emissions in accordance with Kyoto Protocol goals (Newman et al., 2009). Other measures, such as Assembly Bill 32 in California require a reduction in GHGs to 1990 levels by 2020, whereas Seattle's Climate Change Action Plan seeks to achieve net zero emissions by 2050 (Foster et al., 2013).

A number of studies have assessed the embodied energy, carbon footprint and overall environmental sustainability (e.g., includes other environmental impact categories) of WWTPs that are integrated with resource recovery (Lundie et al., 2004; Stokes and Horvath, 2006, 2009; Lyons et al., 2009; Muñoz et al., 2009; Meneses et al., 2010; Muñoz et al., 2010; Pasqualino et al., 2010; de Haas et al., 2011; Mo and Zhang, 2012; Cornejo et al., 2013; Galvin, 2013). However, the majority of these studies do not fully consider the impacts and offsets associated with integrated resource recovery (water reuse, energy recovery, and nutrient recycling). Additionally, various estimation tools have been developed to assess the environmental sustainability of water and wastewater systems (Stokes and Horvath, 2006; Reffold et al., 2008; UKWIR, 2008; Crawford et al., 2011; Johnston, 2011; Corominas et al., 2012; Goel et al., 2012; Tampa Bay Water, 2012; EnviroSim Associates Ltd., 2014). While some of these tools are specific to carbon footprint, other tools have broader capabilities to investigate additional environmental impact categories important to wastewater management (e.g., embodied energy and eutrophication potential). These studies provide designers, managers, and researchers with useful information; however, further research is needed to understand major trends related to the environmental sustainability of WWTPs with resource recovery. Additionally, it is essential to analyze methodologies, frameworks and available tools that calculate the environmental impact of these systems.

The goal of this chapter is to identify the needs for future research and practice that could facilitate accurate comparisons of the environmental sustainability of WWTPs with resource recovery. Previous studies were compared to identify challenges, trends, and major factors impacting the environmental sustainability of wastewater systems implementing water reuse, energy recovery, and/or nutrient recycling. Additionally, environmental sustainability tools for water and wastewater systems were reviewed to identify limitations, challenges, and knowledge gaps. Recommendations are provided to support the development of a more accurate and applicable framework to assess the environmental sustainability of WWTPs with integrated resource recovery. Subsequently, a framework used to investigate global and local environmental impacts of WWTPs with integrated resource recovery at different scales of implementation (Chapter 3) in different contexts (Chapter 4) is presented.

Previous studies have shown that embodied energy, carbon footprint and eutrophication potential are key environmental impact categories for WWTPs integrating resource recovery (Dennison et al., 1998; Hospido et al., 2004; Ortiz et al., 2007; Lyons et al., 2009; Mo and Zhang, 2012; Cornejo et al., 2013). Embodied energy and carbon footprint represent global impacts with both economic and environmental implications (e.g., reducing greenhouse gas emissions is essential for climate change mitigation) (Stokes et al., 2014). Conversely, eutrophication potential represents local impacts important to managing the nitrogen cycle, protecting local water bodies worldwide, and addressing phosphorus scarcity (Mihelcic et al., 2011; NAE, 2012; UNEP, 2014). Collectively, embodied energy, carbon footprint, and eutrophication potential are key environmental sustainability indicators pertinent to the water-energy-carbon-nutrient nexus of wastewater management solutions and resource recovery alternatives and consequently the primary focus of this research.

2.2 The Challenge of Comparing Environmental Impact Results

Whereas life cycle assessment (LCA) tools can be used to investigate a wide range of environmental impact categories (e.g., carcinogens (chloroethylene [C_2H_3Cl] equivalents), ozone depletion (CFC-11 equivalents), respiratory organics (ethylene [C_2H_4] equivalents), aquatic ecotoxicity (triethylene glycol [TEG] water), terrestrial ecotoxicity (TEG soil)), this research focuses on carbon footprint, embodied energy, and eutrophication potential. These impact categories were selected, because they represent key environmental impact categories related to the environmental sustainability of wastewater treatment and resource recovery applications.

2.2.1 The Challenge of Carbon Footprint and Embodied Energy Comparisons

Based on the limited data available in the literature, the estimated carbon footprint of WWTPs that incorporate water reuse and other forms of resource recovery (e.g., energy recovery and nutrient recycling) ranges from 0.1 to 2.4 kg CO_2eq/m^3 (Cornejo et al., 2013). The wide variation in range can be attributed to major impacting factors from representative studies (See Table 7 and Table 8), which include: location, technologies evaluated, life cycle stages considered, parameters considered (i.e., materials, electricity, chemicals, etc.), and estimation methodologies. Implementation scale is also known to be a major factor related to the infrastructure and operation and maintenance cost of WWTPs (EPA, 1978a, 1978b; Fraquelli and Giandrone, 2003; Hopkins et al., 2004); however, no clear trends between implementation scale and associated environmental impact have been demonstrated, highlighting the need for future research in this area.

Location has a large impact on site-specific conditions such as electricity mix, water quality, and geographical conditions (e.g., topography, demographics), leading to changes in environmental impact. For example, various studies show that the electricity mix used for energy production has a significant effect on Scope 2 GHG emissions (Ortiz et al., 2007; Stokes and Horvath, 2009). Similarly, influent water quality and intended level of treatment (e.g., potable versus non-potable) influence technology selection and associated energy consumption (Fine and Hadas, 2012; Stokes and Horvath, 2006; Lyons et al., 2009). Limited studies have investigated how context (location) influences technology selection and the environmental impact of WWTPs with resource recovery.

Table 7. Summary of representative literature evaluating the carbon footprint of WWTPs integrated with resource recovery

Study	Location	Technologies/ Processes	Life Stages	Parameters Considered	Methodology
Tangsubkul et al. (2005)	Australia	CAS with membrane treatment, MBR-RO, waste stabilization ponds	CLS, O&M	Fuel, materials, electricity, chemicals, direct emissions	PLCA, EIO-LCA
Ortiz et al. (2007)	Spain	CAS-Immersed MBR, CAS-External MBR, CAS-Filtration	CLS, O&M, DLS	Materials, delivery, electricity	PLCA
Friedrich et al. (2009)	South Africa	Collection, primary treatment, CAS, flocculation, coagulation, filtration, ozonation, GAC, chlorination	CLS, O&M	Fuel, materials, electricity, chemicals, water offsets	PLCA
Pitterle, 2009	United States	Various (e.g., septic tank with leachfield, CAS with CHP)	CLS, O&M	Fuel, materials, electricity, chemicals, nutrient and energy offsets	PLCA, EIO-LCA
Stillwell and Webber (2010)	United States	Various (e.g., trickling filters, CAS)	O&M	Electricity, water offsets	Electricity and EF
Fine and Hadas (2012)	Israel	Secondary aeration with nitrification/denitrification, clarifiers and deep sand filtration	O&M	Electricity, direct emissions, nutrient and energy offsets	COD, energy and EF
Mo and Zhang (2012)	United States	Primary and secondary treatment, nitrogen removal, post-aeration, and chlorine disinfection	CLS, O&M	Materials, electricity, water offsets, nutrient and energy offsets	EIO-LCA and EF
Shehabi et al. (2012)	United States	Septic tank, sand filter, UV and sedimentation, CAS, disinfection, anaerobic digestion	CLS, O&M	Fuel, materials, electricity, chemicals, water, nutrient, energy offsets	PLCA, EIO-LCA
Cornejo et al. (2013)	Bolivia	Bathrooms, collection, 3-Pond and UASB-Pond Systems	CLS, O&M	Fuel, materials, delivery, electricity, direct emissions, water and energy offsets	PLCA

Note: Most studies include other environmental impact categories in addition to carbon footprint, yet all studies in table incorporate carbon footprint of water reuse systems. CLS – Construction life stage; CAS – Conventional activated sludge; CHP = combined heat and power; COD – Chemical oxygen demand; DLS – Decommission life stage; EF – Emission factor; EIO-LCA – Environmental input/ output life cycle assessment; GAC – Granular activated carbon; MBR – Membrane bioreactor; O&M – Operation and maintenance; PLCA – Process life cycle assessment; RO – Reverse osmosis; RR = resource recovery; UASB – Upflow anaerobic sludge blanket reactor; UV – Ultraviolet; WWTP – Wastewater treatment plant.

Table 8. Summary of representative literature evaluating the carbon footprint of WWTPs integrated with resource recovery and desalination facilities

Study	Location	Technologies/ Processes	Life Stages	Parameters Considered	Methodology
Lundie et al. (2004)	Australia	Filtration, distribution, use, WWTPs, biosolids reuse	CLS, O&M	Materials, electricity, chemicals, transportation, nutrient and energy offsets	PLCA
Stokes and Horvath (2006)	United States	RO versus coagulation, filtration, désinfection	CLS, O&M	Fuel, materials, delivery, electricity, equipment, chemicals	PLCA
Lyons et al. (2009)	United States	RO versus MF/RO, aquifer storage and recovery	CLS, O&M	Materials, electricity, chemicals	PLCA
Muñoz et al. (2009)	Spain	Ozonation (with and without hydrogen peroxide) replacing seawater desalination	O&M	Electricity, chemicals, delivery	PLCA
Pasqualino et al. (2010)	Spain	Collection, grit removal, clarifiers, coagulation, flocculation, filtration, chlorination, and UV replacing desalination	O&M	Materials, delivery, electricity, water and desalinated water offsets	PLCA
Stokes and Horvath (2009)	United States	RO versus filtration and disinfection	CLS, O&M	Fuel, materials, delivery, electricity, equipment, chemicals	PLCA and EIO-LCA
Meneses et al. (2010)	Spain	Chlorination and UV treatment, ozonation, ozonation and hydrogen peroxide, desalination	O&M	Electricity, chemicals, transport of waste, disposal, water offsets	PLCA
Muñoz et al. (2010)	Spain	RO, UV and membranes	CLS, O&M, DLS	Materials, electricity, chemicals	PLCA
de Haas et al. (2011)	Australia	RO and WWTPs producing different water quality	CLS, O&M	Electricity, chemicals, direct emissions, energy, nutrient and water offsets	PLCA

Note: Most studies include other environmental impact categories in addition to carbon footprint, yet all studies in table incorporate carbon footprint of water reuse and desalination systems. CLS – Construction life stage; DLS – Decommission life stage; EIO-LCA – Environmental input/ output life cycle assessment; MF – Microfiltration; O&M – Operation and maintenance; PLCA – Process life cycle assessment; RO – Reverse osmosis; RR = resource recovery; UV – Ultraviolet; WWTP – Wastewater treatment plant.

Topographical conditions can also play a major role in effecting the carbon footprint and embodied energy of these systems. In larger urban areas, wastewater has traditionally been transported through gravity sewers to a centralized wastewater treatment facility located at the lowest elevation in a city (Stokes and Horvath, 2006; Lee et al., 2013). After treatment, pumping energy is often required to transfer water back to end-users through separate distribution infrastructure for reuse, increasing the carbon footprint associated with electricity usage (Scope 2 emissions) and construction materials (Scope 3 emissions). In contrast, less pumping energy may be required in areas with flat topographies. As a result, the estimated carbon footprint and embodied energy is dependent on site-specific topographical conditions such as hills, valleys, plateaus, and waterway locations.

Other factors that impact the estimation of carbon footprint and embodied energy include life stages and parameters considered in the life cycle inventory (i.e., electricity, chemicals, infrastructure, etc.). The literature reviewed includes the operation and maintenance (O&M) stage, but less than half consider the construction stage (Refer back to Tables 7 and 8). Additionally, almost all studies take into account on-site energy usage during O&M that contributes to Scope 2 emissions. However, fewer studies consider the relative contributions from direct process emissions (e.g., CH_4 and N_2O) (Tangsubkul et al., 2005; Foley et al., 2010; de Haas et al., 2011; Fine and Hadas, 2012; Cornejo et al., 2013). Consequently, comparing the environmental impact of systems across different studies poses a challenge when different life cycle stages and parameters are considered. It is therefore imperative to use consistent life stages and parameters when comparing results across systems to ensure the accuracy of the analysis.

Another major challenge to ensuring fair comparison of results across studies is the wide variations in frameworks, methodologies and estimation tools used to analyze the environmental impact. Most of the previous studies used LCA, which often includes supply-chain emissions (Scope 3) associated with material and chemical production (ISO, 2006). The selection of system boundaries in LCA studies changes with the goal and scope of a study, which can lead to difficulties in comparing results. Consequently, a consistent framework with comparable system boundaries is needed to evaluate the impact that context and scale have on the embodied energy and carbon footprint WWTPs with integrated resource recovery.

2.2.2 The Challenge of Eutrophication Potential Comparisons

Similar to carbon footprint and embodied energy, comparisons of eutrophication potential results from previous studies are difficult because of changes in location, system boundaries, methodologies, and limited studies exploring eutrophication trends in depth. For example, TRACI and ReCiPE methods apply different methods to calculate eutrophication potential, so the results for the same inputs differ (Pre Consultants, 2014). Furthermore, life cycle assessment studies often explore a wide range of environmental impact categories. This is beneficial to gaining an understanding of the overall environmental impact; however, this approach often does not include enough in-depth information to consider how scale and context impact eutrophication potential.

Eutrophication potential from WWTPs implementing resource recovery ranges from 0.03 to 1.00 g PO_4eq/m^3 (Meneses et al., 2010; Pasqualino et al., 2010) and are largely dependent on local conditions, technology selected, treatment efficiency, and effluent water quality. The high end of this range comes from the replacement of potable water from WWTP and tertiary treatment where the low end of this range comes from agricultural reuse from tertiary treatment only. These studies only consider indirect sources of eutrophication (e.g., NO_x from electricity), where direct emissions are excluded. In this case, the more comprehensive system boundary considered for potable water replacement leads to a higher eutrophication potential due to a larger contribution from indirect sources of eutrophication.

Consequently, further research is needed to understand how context and scale influence eutrophication potential and trade-offs associated with varying technologies. Previous studies have observed that environmental problem shifting may occur between global and local environmental impacts. For example, Foley et al. (2010) observed that higher levels of nutrient removal require more electricity and infrastructure, leading to a reduction in nutrient pollution, but an increase in energy consumption and associated carbon emissions. This represents a trade-off where solving environmental problems related to local impacts (e.g., reducing eutrophication in local water bodies) can lead environmental problem shifting at the global scale (e.g., increased embodied energy and carbon footprint). Consequently, the trade-offs associated with embodied energy, carbon footprint, and eutrophication potential merit further investigation.

2.3 Environmental Sustainability Trends for WWTPs with Resource Recovery

2.3.1 Trends of Carbon Footprint and Embodied Energy: Global Impacts

Carbon footprint and embodied energy are closely related, where direct energy (e.g., electricity) and indirect energy (i.e., materials, chemicals, etc.) contribute Scope 2 and 3 greenhouse gas (GHGs) emissions of WWTPs with integrated resource recovery, respectively. Consequently, the discussion in this section focuses primarily on carbon footprint, yet both of these impact categories (e.g., carbon footprint and embodied energy) represent major global impacts of the systems investigated. Direct (Scope 1) emissions from individual GHGs are also discussed, where carbon footprint is defined as the sum of individual greenhouse gas emissions, including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Methane and nitrous oxide are expressed in carbon dioxide equivalents (CO_2eq) by converting CH_4 and N_2O emissions using their global warming potential (IPCC, 2006; Mihelcic et al., 2013). Both methane and nitrous oxide are important greenhouse gases for WWTPs with large 100-year global warming potentials at 25 and 298, respectively (IPCC, 2007).

Currently, more than 50% of the groundwater supplies used worldwide are over-drafted, placing pressure on aquifers used for human activities (Brown, 2011; Schroeder et al., 2012). Both water reuse and desalination represent two major water provision alternatives to conventional water supplies (e.g., surface water, groundwater). Despite the intrinsic challenges in comparing the carbon footprint results from various studies, the carbon footprint of desalination systems was generally found to be higher than water reuse systems (Lundie et al., 2004; Stokes and Horvath, 2006, 2009; Lyons et al., 2009; Muñoz et al., 2009; de Haas et al., 2011). Reverse osmosis (RO) technologies were found to have lower CO_2 emissions than thermal desalination technologies and the estimated carbon footprint of seawater RO desalination (0.4–6.7 kg CO_2eq/m^3) is generally larger than brackish water RO desalination (0.4–2.5 kg CO_2eq/m^3) and water reuse systems (0.1–2.4 kg CO_2eq/m^3), highlight the importance of water reuse as a sustainable alternative water supply.

Various examples in the literature highlight that WWTPs that employ water reuse and other forms of resource recovery are more environmentally sustainable than desalination. For example, Stokes and Horvath (2006) found that a seawater desalination facility with flocculation, filtration, RO, and disinfection processes had a carbon footprint three times greater than a water reclamation system with coagulation, filtration and disinfection steps. In that study, seawater was treated to potable standards for potable water consumption while reclaimed water was treated to replace potable water used for irrigation and other non-potable reuse applications. Another study found the carbon footprint of certain tertiary technologies for water reuse (e.g., ozone or ozone peroxide) was 85% less than seawater RO desalination (Muñoz et al., 2009). Expanding on the work of Muñoz et al. (2009), Meneses et al. (2010) found that the carbon footprint of UV and chlorination disinfection options for water reuse were comparable to ozone and ozone peroxide. Given the environmental benefits to water reuse, various utilities have turned to reclaimed water to replace potable water supplies used for non-potable purposes. Additionally, there are generally economic advantages to indirect potable reuse (820-2,000

$/acre-foot) and non-potable reuse (320-1,960 $/acre-foot), compared to seawater desalination (1,500-2,330 $/acre-foot) (Raucher and Tchobanoglous, 2014).

Studies on WWTPs (Stokes and Horvath, 2006; Ortiz et al., 2007; Friedrich et al., 2009; Lyons et al., 2009; Pasqualino et al., 2010; de Haas et al., 2011) generally found that energy consumption is a dominant factor contributing approximately 68 to 92% of the carbon footprint (Tangsubkul et al., 2005; Stokes and Horvath, 2009). Many studies confirmed that aeration using conventional activated sludge (CAS) during wastewater treatment led to high electricity consumption (Friedrich et al., 2009; Pasqualino et al., 2010; Zhang et al., 2010) and consequently high Scope 2 emissions during the operation phase, as expected. Conversely, methane emissions were found to be a dominant contributor (approximately 58 to 69%) to the overall carbon footprint of systems that implement natural wastewater treatment methods, such as waste stabilization ponds (Tangsubkul et al., 2005; Cornejo et al., 2013). This large contribution from CH_4 highlights the importance of direct emissions (Scope 1), particularly for natural wastewater treatment technologies.

Generally, the carbon footprint of secondary treatment is higher than the carbon footprint of tertiary treatment using filtration and disinfection processes for reuse. For example, Friedrich et al. (2009) found that conventional activated sludge (CAS) contributed three times more CO_2 than a tertiary treatment train (e.g., coagulation, sand/anthracite filtration, ozonation, granular activated carbon (GAC) and chlorination), where 90% of the CO_2 emissions were associated with electricity consumption. In another study, Pasqualino et al. (2010) found that the carbon footprint of primary, secondary and sludge treatment (0.83 kg CO_2eq/m^3) was greater than a tertiary treatment train including coagulation, flocculation, chlorination, sand filtration and UV disinfection (0.16 kg CO_2eq/m^3).

The level of treatment has also been found to impact the carbon footprint and associated embodied energy results in previous studies (Foley et al., 2010). This trend is demonstrated in Table 9, where the carbon footprint increases as treatment level increases for varying end-use applications. Consequently, secondary and tertiary treatment suitable for indirect potable reuse has a higher carbon footprint than secondary treatment suitable for non-food crop irrigation, as expected. However, this increased level of treatment for nutrient removal leads to trade-offs, in which embodied energy and carbon footprint increase, while eutrophication potential decreases (Foley et al., 2010).

Table 9. Carbon footprint and carbon dioxide emissions per m^3 of produced water for water reuse systems at different treatment levels

End-Use	Recommended Treatment Level	Carbon Footprint (kg CO_2eq/m^3)	Carbon Dioxide Emissions (kg CO_2/m^3)	Remarks
No use recommended	Primary	0.11-0.16	-	Primary treatment is generally lower than secondary and tertiary treatment
Non-food crop irrigation[a]	Secondary	0.30-2.0	0.13-0.69	For CO_2 emissions, low point from Norwegian electricity mix, high value from average European electricity mix, average airborne emissions
Indirect potable reuse[b]	Secondary and tertiary	0.6-2.4	0.14-0.98	For carbon footprint, low value is for demand-driven advanced treatment and high value is advanced treatment for 100% of the wastewater effluent

[a] Includes restricted landscape irrigation, surface irrigation of orchards and vineyards, groundwater recharge of non-potable aquifer, stream augmentation, industrial cooling (Mo and Zhang, 2013). [b] Includes landscape irrigation, urban reuse, food crop irrigation, indirect potable reuse (Mo and Zhang, 2013). Sources: Lundie et al., 2004; Tangsubkul et al., 2005; Ortiz et al., 2007; Friedrich et al., 2009; Lyons et al., 2009; Pasqualino et al., 2010; de Haas et al., 2011; Fine and Hadas, 2012; Mo and Zhang, 2012; Cornejo et al., 2013.

Limited research has been conducted on the carbon footprint of technologies used to achieve specific trace constituent removal for direct potable reuse (Leverenz et al., 2011). However, Sobhani and Rosso (2011) studied the contribution of an advanced oxidation process (AOP) in treating N-Nitrosodimethylamine, a possible cancer-causing agent, to the overall energy and carbon footprints of the indirect potable reuse system in Orange County, California.

It was estimated that influent pumping, primary treatment, secondary treatment, micro-filtration, AOP, and RO contributed 3%, 4%, 16%, 21%, 7%, and 49% of the total energy footprint, respectively. This suggests that RO and AOP contribute approximately half of the total energy consumption. Additionally, the study highlighted that there is a difference between technologies required for non-potable reuse (e.g., landscaping and irrigation) as opposed to potable reuse, which typically involves advanced treatment including RO and AOP.

2.3.2 Trends of Eutrophication Potential: Local Impacts

Eutrophication occurs when nutrients (e.g., nitrogen and phosphorus) in water bodies cause an increase in plant and algal growth, thereby depleting oxygen levels and the health of aquatic ecosystems (Pasqualino et al., 2010). Previous studies have shown that nutrients in effluent discharges and biosolids land application from WWTPs are large contributor to eutrophication (Dennison et al., 1998; Hospido et al., 2004; Foley et al., 2010). Additional sources of eutrophication potential come from indirect sources, such as NO_x emissions from transportation, electricity production, and chemical production (Lundie et al., 2004).

Reducing nutrient loads discharged to water bodies can lead to reductions in eutrophication potential. For example, Hospido et al. (2004) found that implementing nitrification-denitrification with biological treatment for reductions of nutrients and organic matter can reduced eutrophication by 54-58%; however, higher energy requirements needed for additional treatment led to increases in carbon footprint. Another study found that implementation of a greenfield scenario (e.g., appliances for water efficiency, rainwater catchment, household primary treatment and nutrient removal at neighborhood scale, water reuse for irrigation and regional treatment of biosolids) led to reductions in eutrophication potential by a factor of 10 (Lundie et al., 2004). In addition, Meneses et al. (2010) found that fluctuations in effluent nitrogen concentrations had a large impact on eutrophication potential results, where a 10% change in effluent nitrogen content can lead to a 37% change in eutrophication potential. These studies highlight the importance of nutrient removal and nutrient recycling as a means to reduce the eutrophication potential from WWTPs.

Recycling nutrients through water reuse and land application of biosolids can therefore reduce the impacts of eutrophication by minimizing excess levels of nutrient discharged directly to water bodies from treated effluents (Pasqualino et al., 2010). The benefits of water and nutrient reutilization, growing urban water demands, water scarcity, efficient resource utilization, and the protection of human and ecosystem health are all drivers for water reuse (EPA, 2012b). In the United States, Florida and California are national leaders in water reuse. In 2013, Florida reclaimed 719 million mgd of water, representing approximately 66% of the state's permitted domestic WWTP capacity (FDEP, 2014a). Reduction of eutrophication potential in Florida is of particular importance because nutrient pollution can negatively impact human health, the environment, and freshwater and seawater based tourism (EPA, 2015a). Whereas eutrophication potential is an important environmental impact category for wastewater management, limited research has been conducted on the influence of context and scale on eutrophication potential from WWTPs that recover nutrients and water.

2.3.3 Trends of WWTPs with Integrated Resource Recovery Offsets

Some studies incorporate the benefits associated with resource recovery as credits in the carbon footprint or embodied energy calculations (Lundie et al., 2004; Meneses et al., 2010; Pasqualino et al., 2010; Stillwell and Webber, 2010; Fine and Hadas, 2012; Mo and Zhang, 2012). This is due to the fact that potential carbon footprint and/or embodied energy offsets are provided through water reclamation (e.g., offsets energy used to treat potable water), nutrient recovery (e.g., offsets synthetic fertilizers), and energy recovery (e.g., offsets electricity from grid) activities. Few studies fully incorporate offsets from integrating water reuse, nutrient recycling and energy recovery collectively (Lundie et al., 2004; Pitterle, 2009; Mo and Zhang, 2012); although most studies consider at least two of the three resource recovery strategies (Tillman et al., 1998; Meneses et al., 2010; Fine and Hadas, 2012; Galvin, 2013). Although adoption of individual resource recovery strategies can lead to beneficial offsets, integrating all three strategies leads to the greatest offset potential. For example, Mo and Zhang (2012) found that integrated resource recovery can offset all of the operational energy of a WWTP in Tampa.

Water reuse is known to mitigate the embodied energy (37-41% offset) and carbon footprint (36-40% offset) of WWTPs (Mo and Zhang, 2012). For example, one study found that water reuse implemented at 12% of the total water demand in the state of Texas could lead to a net energy savings of 73-310 million kWh per year and 0.04-0.16 million metric tons of CO_2 offset annually (Stillwell and Webber, 2010). The mitigation potential of water reuse, however, is largely dependent on the existing quality and desired level of treatment of the water that is being replaced. For example, Pasqualino et al. (2010) and Shehabi et al. (2012) found that desalinated water replacement has a higher energy and carbon mitigation potential than potable water replacement, because desalination is more energy-intensive. In contrast, when replacing untreated surface water Pitterle (2009) found that water reuse had no benefit to offset embodied energy and carbon footprint. Consequently, replacing higher quality water (e.g., potable water), leads to greater energy and carbon offsets than replacing low quality water (e.g., non-potable water). Since treating water to a higher level requires more energy and resource inputs, the carbon footprint offset potential of reclaimed water typically increases with higher-value end uses (e.g., replacing high-purity water for industrial processes has a higher offset potential than agricultural reuse) (Pasqualino et al., 2010; Shehabi et al., 2012; Tong et al., 2013).

Fertilizers avoided due to nutrient recycling from reclaimed water and biosolids can also lead to offsets of embodied energy and carbon footprint. Typically, nutrient recycling leads to minor energy offsets (0.1-2% of the total energy) when replacing synthetic fertilizers (Tillman et al., 1998; Pitterle, 2009; Mo and Zhang, 2012). Most of the previous studies considering fertilizer offsets have focused on nutrient recovery from the land applied biosolids. In contrast, other studies have focused on phosphorus and nitrogen recovery from urine (Tillman et al., 1998; Lundin et al., 2000; Mihelcic et al., 2011), whereas few studies have investigated nutrient recycling from reclaimed water. An estimated 22% of the global phosphorus demand could be met through nutrient recycling from urine and feces, while addressing phosphorus scarcity (Mihelcic et al., 2011). Reclaiming nutrient-rich water for beneficial reuse can lead to reductions in eutrophication potential, embodied energy, and carbon footprint (Lundie et al., 2004).

Only one study (Galvin, 2013) investigated life cycle impacts of nutrient recycling from decentralized wastewater treatment in a Latin American context. Galvin (2013) found the nutrient recycling and associated fertilizer replacement from household composting latrines and biodigester latrines in rural Peru can lead to a net energy balance. Furthermore, this study found that nutrient recycling was more effective than energy recovery in offsetting energy from the biodigester latrine due to the high fertilizer value associated with animal manure. This highlights how differences in context, technologies, and nutrient values of different waste types, can impact the environmental sustainability of wastewater treatment systems. Galvin (2013) found that biodigester latrines in rural Peru can mitigate up to 62.4% of energy through nutrient recovery alone.

Previous literature has also shown that energy recovery can lead to reductions in embodied energy ranging from 4 to 30.6 percent in United States and Sweden (Lundie et al., 2004; Pitterle, 2009; Mo and Zhang, 2012; Shehabi et al., 2012). Energy recovery potential varies with system size and organic load, where larger systems treating higher organic loads can generate more energy. Furthermore, CHPs have been reported to only be cost-effective for WWTPs above 5 mgd (EPA, 2007; Mo and Zhang, 2013). Limited research has been conducted on the life cycle impacts of smaller energy recovery applications (less than 5 mgd) from anaerobic treatment processes in developing regions. For example, upflow anaerobic sludge blanket reactors (UASBs) with biogas recovery potential, commonly used in South Asia and Latin America (Lettinga, 2010; Verbyla et al., 2013) have largely been ignored in previous studies. Verbyla et al. (2013) found that theoretically a UASB system in rural Bolivia could produce 10-13 kg CH_4/day, representing 500-650 MJ/day of energy. Galvin (2013) investigated the life cycle impacts of decentralized biodigester latrines treating animal waste in rural Peru and found that the natural gas use avoided through energy recovery can lead to net energy balance. Household biogas digesters can be beneficial to offsetting GHGs associated with burning firewood or fossil fuels; however, these systems are often poorly managed leading to unintended methane releases that can contribute more GHGs than conventional fuels sources in the developing world (Bruun et al., 2014). Consequently, larger community scale energy recovery applications in the developing world merit further investigation.

2.4 Environmental Sustainability Tools for Water and Wastewater System Evaluation

2.4.1 Availability and Applicability

Sixteen available emission tools were reviewed with varying levels of applicability to WWTPs with resource recovery. The tools also varied in calculation capabilities. While some tools focus on a wide range of impact categories (e.g., process LCA) and others focus specifically on carbon footprint as an environmental sustainability metric (e.g., Tampa Bay Water). The tool type (e.g., software, MS-Excel, web-based), availability (e.g., commercial, public, upon request), and source of the various tools are highlighted in Table 10. The different tools may be classified as (1) process LCA tools, (2) hybrid LCA tools, (3) specific tools, and (4) other related tools. Eight out of the sixteen available tools are software-based, six are MS-Excel spreadsheets, and two are web-based. Additionally, eight out of the sixteen tools are commercially sold, five are available on request, and three are publicly available online.

Generally, the application of process LCA tools involves the use of process-based inventories to calculate the environmental impact of any system. This methodology is beneficial in terms of flexibility and analysis of specific processes, but requires a consistent framework to analyze specific systems (e.g., WWTPs with integrated resource recovery). In contrast, hybrid LCA tools and the UK Environment Agency tool were specifically designed to estimate the environmental impact of water, wastewater, water reuse and desalination facilities. Hybrid LCA tools used process-based inventories and economic input output life cycle assessment (EIO-LCA) for carbon footprint estimates. Consequently, it is important to draw from a wide range of tools to understand key life stages, parameters considered, and input data requirements for WWTPs with integrated resource recovery.

The hybrid LCA tools are specifically designed to assess facilities in the United States, whereas the UK Environment Agency tool is specific to facilities in the United Kingdom. Specific tools (e.g., Tampa Bay Water and Johnston tools) and the Carbon Accounting Workbook (UK) are applicable to water facilities. However, the Tampa Bay Water tool is also applicable to desalination facilities and the Johnston tool contains some disinfection and desalination processes that could be useful for estimating the carbon footprint of water reuse or desalination facilities. The remaining tools are applicable to wastewater treatment facilities and therefore, contain attributes that are useful for estimating the carbon footprint of WWTPs with resource recovery.

Table 10. Description of available carbon footprint and environmental sustainability tools related to wastewater treatment facilities with resource recovery and desalination systems

Type	Description of Methodology	Estimation Tool	Tool Format	Available	Applicable	Source
Process LCA-based tools	Use process-based inventory over life cycle[a]	SimaPro	Software	Commercial	Varies, any product or process	www.pre.nl
		Gabi	Software	Commercial		www.gabi-software.com
		SiSOSTAQUA	Software	Commercial		www.simpple.com
Hybrid LCA-based tools	Use both process-based and input-output based inventory over life cycle[b]	WEST	MS-Excel	Upon request	Water, water reuse, desalination	Dr. Jennifer Stokes at ucbwaterlca@gmail.com
		WWEST	MS-Excel	Upon request	Wastewater	Dr. Jennifer Stokes at ucbwaterlca@gmail.com
		WESTWeb	Web-based	Public	Water, water reuse, desalination, wastewater	west.berkeley.edu
Specific tools	Uses input parameters specific to utility over O&M	Tampa Bay Water[c]	MS-Excel	Upon request	Water and desalination	www.tampabaywater.org
		Johnston Tool[d]	MS-Excel	Upon request	Water	Dr. Tanju Karafil at tkaranf@clemson.edu
Other related tools	NOT specifically used to estimate emissions from water reuse or desalination facilities, but contain aspects that are applicable	CHEApet[e]	Web-based	Public	Wastewater	cheapet.werf.org
		UK Environment Agency tool[f]	MS-Excel	Upon request	Water supply, water reuse, desalination	Environment Agency at enquiries@environment-agency.gov.uk
		Bridle and BSM2G tool[g]	Software	Public	Wastewater	Author Lluis Corominas at lcorominas@icra.cat
		System Dynamics[h]	Software	Commercial	Varies	www.iseesystems.com
		GPS-X[i]	Software	Commercial	Wastewater	www.hydromantis.com/GPS-X.html
		Carbon Accounting Workbook, 5th version[j]	MS-Excel	Commercial	Water	www.ukwir.org
		mCO2[k]	Software	Commercial	Wastewater	www.mwhglobal.com
		BioWin 4.0[l]	Software	Commercial	Wastewater	www.envirosim.com

Sources: [a]ISO, 2006; [b]Stokes and Horvath, 2006, 2011a, 2011b; [c]Tampa Bay Water, 2012; [d]Johnston, 2011; [e]Crawford et al., 2011; [f]Reffold et al., 2008; [g]Corominas et al., 2012; [h]Shrestha et al., 2011; [i]Goel et al., 2012; [j]UKWIR, 2008; [k]MWH, 2012; [l]EnviroSim Associates Ltd., 2014.

A major benefit of process-based LCA is that results are specific to material, energy, and waste input processes considered. Therefore, results can be expressed to compare specific processes to gain a more in depth understanding of specific trends. Process LCA, therefore allows flexibility in assessing the life cycle impacts of systems in different settings and varying scales. In contrast, the hybrid LCA models reviewed (e.g., WWEST, WEST) require the use of economic input output life cycle assessment (EIO-LCA). EIO-LCA uses economic input output matrices of specific countries (e.g., United States, China) to estimate the environmental impact of a system based on interactions between economic sectors (Green Design Institute, 2015). EIO-LCA has economic input-output tables for the United States and China, but does not include economic input output data from countries in Latin America (e.g., Bolivia). This limits the applicability of WWEST and EIO-LCA in developing world settings, though EIO-LCA has been applied to investigate the embodied energy of varying water provision strategies in the developing world (Held et al., 2013). It is important to note that the input data collection from WWEST provides one of the most comprehensive frameworks and can be applied to a process-based LCA model. Consequently, this research uses a process-based LCA model to evaluate the embodied energy, carbon footprint, and eutrophication potential of WWTPs with integrated resource recovery in developed and developing world settings, drawing from other tools to create a comprehensive framework.

2.4.2 Knowledge Gaps, Limitations, and Challenges of Existing Tools

Knowledge gaps and key challenges associated with existing frameworks and tools applicable to WWTPs with resource recovery are summarized in Table 11. Further research in these critical areas is needed to develop a comprehensive framework that enables accurate estimations of key environmental impact categories (e.g., carbon footprint). Gaps, limitations and challenges are discussed in the following sections as they relate to parameters and life stages considered, input data, output data, and additional useful attributes.

Table 11. Knowledge gaps, limitations, and challenges of environmental sustainability tools for wastewater treatment systems integrated with resource recovery

Tool Aspect	Knowledge Gaps/Limitations/Key Challenges
Parameters and Life Stages Considered	-Knowledge Gap: Contribution of direct emissions from WWTPs. -Knowledge Gap: Emissions of membranes production, renewal and disposal and brine disposal. -Knowledge Gap: Appropriate allocation methods to account for resource recovery. -Key Challenge: Reaching consensus on the appropriate parameters and life stages to consider.
Input Data	-Limitation: Availability of input data for existing tools. -Key Challenge: Develop model with enough detailed data to determine critical areas for GHG mitigation.
Output Data	-Limitation: Lack of separation of carbon footprint and embodied energy by unit process. -Limitation: Lack of separation of carbon footprint by scope 1, 2, and 3 emissions. -Key Challenge: Conducting comparable estimations for each unit process.
Additional Useful Attributes	-Limitation: User-friendly, regionally-transferable tool widely used -Limitation: Methods for model calibration, validation and/or sensitivity analysis embedded in tool. -Key Challenge: Integration of robust and accurate tool, which combines beneficial attributes.

2.4.2.1 Life Stages and Parameters Considered

Differences in results arise from differences in specific life stages, parameters, and system boundaries considered in the environmental sustainability tools. The system boundary selection is important because it has previously been shown to affect the environmental impact of systems (Lundin et al., 2000). Therefore, it is critical to the accuracy of the study to select a consistent system boundary when comparing different wastewater treatment and resource recovery systems. Figure 4 shows an example of the variation in system boundaries selected in previous LCA studies of wastewater treatment systems. Energy recovery and water reuse are not considered in the system boundaries of this study.

A summary of parameters considered by hybrid LCA tools and specific tools is provided in Table 12. This table highlights that the carbon footprint from operational electricity consumption and the associated electricity mix are the only parameters considered by both hybrid LCA and specific tools. Although operational electricity consumption was found to be a dominant contributor to the carbon footprint in previous studies (Ortiz et al., 2007; Friedrich et al., 2009), other emission sources can also be important. The hybrid LCA framework allows users to estimate impacts associated with construction and operation and maintenance stages, whereas specific tools focus solely on the operational life stage. Despite the dominance of operation phase emissions, studies integrating natural wastewater treatment technologies (e.g., waste stabilization ponds) found that the construction phase was important, accounting for 25-42% of the total carbon footprint (Tangsubkul et al., 2005; Cornejo et al., 2013).

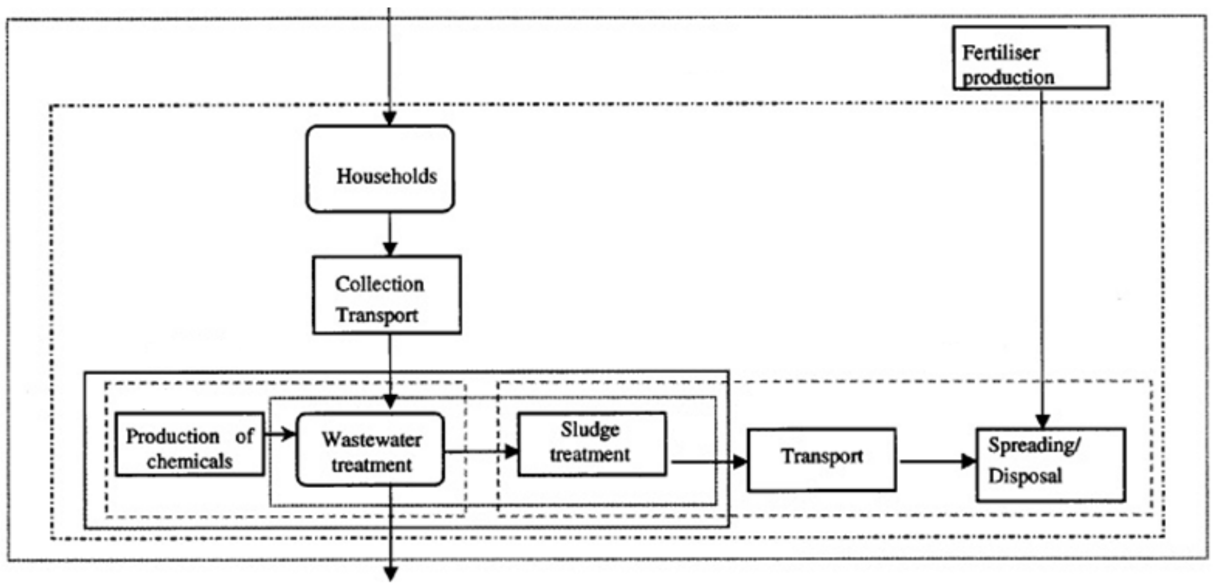

Figure 4. Variation in system boundaries for different LCA studies. Reprinted (adapted) with permission from Lundin et al. (2000). Copyright 2000 American Chemical Society.

The Johnston tool and hybrid LCA tools consider a more complete set of parameters. For example, the Wastewater Energy Sustainability Tool (WWEST) and Water Energy Sustainability Tool web version (WESTWeb) can estimate direct process emissions (e.g., CH_4 and N_2O) from various wastewater treatment processes based on water quality data and population served. Direct process emissions can play a significant role in carbon footprint mitigation efforts since they can be directly controlled through process modifications (Stokes and Horvath, 2010; de

Haas et al., 2011). Further research is needed to quantify the direct process emissions (e.g., fugitive CH_4 and N_2O) and the carbon footprint reduction due to control technologies. The Johnston tool, WWEST, and WESTWeb also include several process-specific carbon footprint estimates from relevant materials and equipment (e.g., filter media, membranes, and blowers). This enables the identification of carbon intensive processes, which can enhance mitigation efforts.

Table 12. Parameters considered by hybrid LCA and specific tools that contribute to the carbon footprint and environmental impact

Parameters Considered	Hybrid LCA Tools			Specific Tools	
	WEST[a]	WWEST[a]	WESTWeb[a]	Johnston Tool[b]	Tampa Bay Water Tool[c]
Material production	X	X	X		
Material delivery	X	X			
Electricity consumption	X	X	X	X	X
Electricity mix	X	X	X	X	X
Fuel use (on-site and fleet vehicles)	X	X	X	X	
Sludge disposal	X	X	X	X[d]	
Chemical production	X	X	X	X	
Direct process emissions		X[e]	X[e]	X[d]	
Process equipment	X[f]	X[f]	X[f]	X[g]	
Disinfection processes	X[f]	X[f]	X[f]	X[g]	

X = included. Sources: [a]Johnston, 2011; [b]Stokes and Horvath, 2011a, 2011b; [c]Tampa Bay Water, 2012. [d]Direct emission factors for ozone generation, GAC, reservoirs, and sludge disposal from potable water production, not applicable to water reuse or desalination; [e]Direct emission for various wastewater treatment processes; [f]Includes filter media (sand, gravel, anthracite, or other coal product), membranes, pumps, fans/blowers, motors and generators, turbines, metal tanks, UV lamps/lights, other industrial equipment, electrical, controls; [g]Utilities can estimate energy consumption from mixers, flocculators, settlers, DAF, filtration, MF/UF, UV, ozone, hypochlorite, decarbonators, RO, and thermal desalination by entering the average flow rate.

The wastewater energy sustainability tool (WWEST) developed by Stokes and Horvath (2010) contains one of the most comprehensive system boundaries for wastewater treatment including the following parameters: material production, material delivery, equipment operation, energy production, sludge disposal and direct emissions. This framework considers the collection, treatment, and distribution of wastewater over the construction and operation phases.

Ideally, all of these activities would be included in the system boundary of a wastewater treatment system; however, data may not be available for all of these activities. Data collection is especially a challenge in the developing world, where data availability is limited (Held, 2013).

The WWEST structure played a central role in aiding the framework development and data collection of this research. This structure should; however, be extended to included specific modules that capture the mitigation potential of water reuse, energy recovery, and nutrient recycling. This model already includes some resource recovery features, such as gas recovery for anaerobic digestion and the quantification of fertilizers as co-products. However, it is not specifically designed to include water reuse or other unique integrated resource recovery strategies. Enhancements of the WWEST structure are the inclusion of:

- Water reclamation as a co-product for replacement of different types of water (e.g., replacement of potable water, replacement of desalinated water, replacement of surface water)
- Nutrient benefit of reclaimed water used for irrigation for different types of end-uses (e.g., agricultural irrigation, urban reuse, etc.)

2.4.2.2 Input Data

A major difference between the hybrid LCA tools and the specific tools is the amount of input data required for a comprehensive analysis. A large amount of data is required to conduct a comprehensive analysis using hybrid LCA tools. Users are not required to enter all of the inputs; however, the arbitrary selection of default data inputs could lead to inaccurate estimations. Additionally, some facilities may not have or collect sufficient input data required by the hybrid LCA tools. The lack of input data collected in practice is thus a limitation to the successful implementation of the hybrid LCA tools. In contrast, the specific tools require fewer inputs than the hybrid LCA tools, since they focus only on emissions associated with the operational life stage. Fewer inputs could be beneficial to facilitate widespread adoption and provide water utility decision makers an easy-to-use tool for evaluation of carbon footprint.

To evaluate the differences of available tools, two were compared using data from a previous study (Stokes and Horvath, 2009) as seen in Figure 5. The Tampa Bay Water tool represents the simplest available tool requiring minimal data inputs (e.g., electricity consumption, electricity mix), whereas WESTWeb (Water Energy Sustainability Tool, web version) represents a more sophisticated tool requiring extensive data inputs (e.g., material production, chemical usage, fuel usage, electricity consumption, electricity mix).

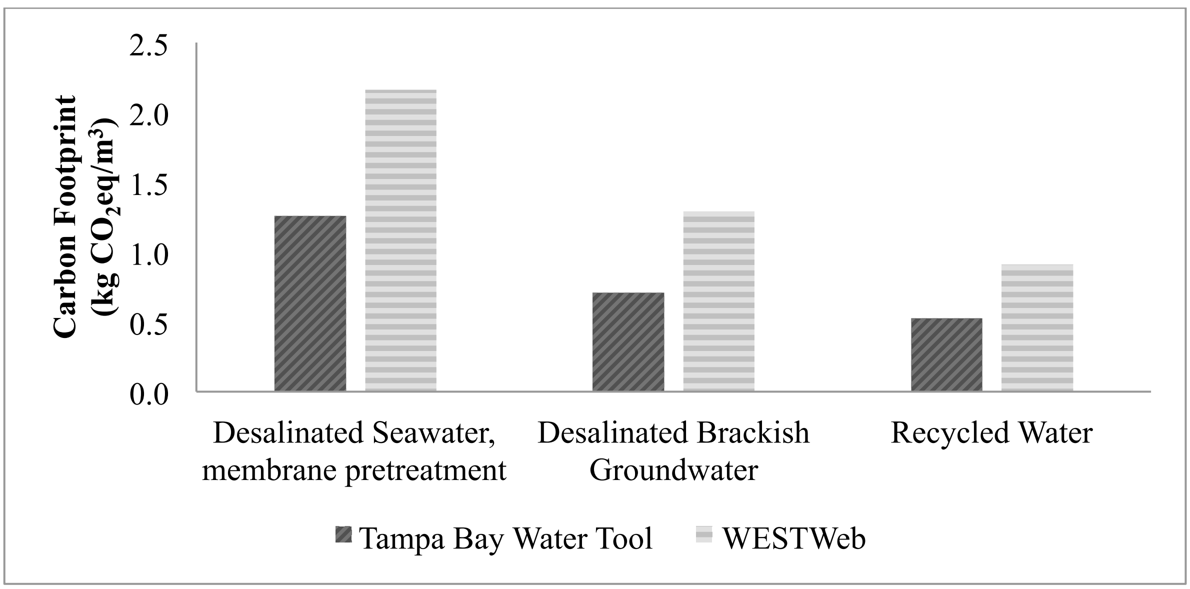

Figure 5. Comparison of carbon footprint estimate using Tampa Bay Water and WESTWeb tools

The estimated carbon footprint for three different facilities assessed (capacity of 26.1 mgd): a seawater desalination facility, a brackish groundwater desalination facility, and a water reuse facility. These estimations fall within ranges reported previously for seawater RO desalination, brackish water RO desalination, and water reuse. The carbon footprint per cubic meter of produced water from the Tampa Bay Water tool accounts for 55-58% of the WESTWeb estimate. The difference in estimations demonstrates that the Tampa Bay Water tool underestimates life cycle impacts included in the more comprehensive hybrid LCA tool. This highlights the importance of considering parameters and life stages included in the more comprehensive hybrid LCA framework. Other tools, such as process LCA could also used to consider a more diverse set of parameters. This would allow for a more comprehensive and holistic analysis than only considering operational electricity consumption and electricity mix.

2.4.2.3 Output Data

A limitation for most tools is the lack of distinction between Scope 1, 2, and 3 carbon footprint results. The Tampa Bay Water tool, for example, only presents Scope 2 results from electricity consumption, whereas the hybrid LCA tools present all Scope 1, 2, and 3 results collectively. The Johnston tool is the only framework that presents carbon footprint results as Scope 1 (direct), Scope 2 (indirect), and Scope 3 (other indirect) emissions for water treatment estimates. Enhancements to outputs of existing frameworks include:

- The separation of unit processes to enable the identification high impact areas and comparisons
- The categorization of carbon footprint results expressed as direct Scope 1 emissions (e.g., CH_4 and N_2O process emissions), indirect Scope 2 emissions (e.g., electricity), and other indirect Scope 3 emissions (e.g., materials), consistent with published protocols for GHG classifications (e.g., Local Government Operations Protocol and WRI/WBCSD GHG Protocol Corporate Standard).

Existing and voluntary carbon footprint reporting programs include Scope 1 and Scope 2 emissions, as will potential future regulations or cap-and-trade programs (Huxley et al., 2009). Similarly, separation of direct and indirect emissions from other impact categories (e.g., embodied energy and eutrophication potential) is recommended to understand critical mitigation areas of specific processes.

2.5 Sustainability Framework for WWTP with Integrated Resource Recovery

Sustainability frameworks should be rooted in the core definition of sustainable development. A commonly used definition of sustainable development is: "development that meets the needs of the present without compromising the ability of future generations to meet their own needs." (IISD, 2013). This includes the three pillars of sustainability that emphasize social, environmental and economic well-being (Mihelcic et al., 2003; Anastas, 2012). Although the current research focuses solely on environmental sustainability, future works should incorporate broader definitions of social and economic concerns in their operational framework. To design the current framework, the following characteristics recommended by the Environmental Protection Agency's (EPA) operational frameworks for sustainability were incorporated (NRC, 2011):

- Clarity and transparency
- Practical implementation
- Measurable goals and objectives that can be reported to the public
- Flexibility to adapt to scientific, technical, and economic developments over time
- Consistent with EPA's current risk management paradigm
- Facilitates decision-making to protect human health and the environment

2.5.1 Methodology

Life cycle assessment (LCA) is the application of life cycle thinking to evaluate environmental impacts of a system. This quantitative tool estimates the environmental impact of a system over its life, including raw material extraction, construction, operation, reuse and end-of-life phases. The following steps are used to conduct a life cycle assessment, in accordance with ISO standard 14040 (ISO, 2006):

- Goal and scope definition
- Inventory Analysis
- Impact Assessment
- Interpretation

The goal and scope define the goal of the study, the system boundary, and the functional unit. While the system boundary defines what life stages and phases are included in a system, the functional unit provides a unit of comparison for different systems based on their function over the life cycle. The inventory analysis compiles material, chemical and energy inputs, as well as relevant output emissions. The impact assessment evaluates this inventory to calculate selected environmental impact categories. Interpretation of results is conducted throughout the LCA, which often includes a sensitivity analysis or uncertainty analysis. Sensitivity and uncertainty analyses are used to evaluate the sensitivity and uncertainty of the results, respectively.

2.5.1.1 Goal and Scope Definition

The goal and scope definition designate the goal of the study, functional unit, and system boundaries. The goal of this research is to evaluate the environmental sustainability of existing wastewater treatment systems with integrated resource recovery in Bolivia and the United States. Embodied energy, carbon footprint, and eutrophication potential were selected as sustainability indicators. The system boundaries of all systems include collection, treatment and distribution of wastewater over the construction and operation life stages (See Table 13). Additionally, the impact and offset of water reuse, energy recovery, and nutrient recycling are considered where applicable. A functional unit of 1 cubic meter of treated wastewater was selected over a 20 year time period, the typical lifespan of WWTPs. This is a conservative estimate since the lifespan of water infrastructure prior to replacement or rehabilitation is 20-50 years (EPA, 2012c). The functional unit is based on the primary function of the system, which is to treat water over its useful life.

Table 13. System boundary and life stages considered in current framework used to investigate the environmental sustainability of wastewater treatment systems with integrated resource recovery

		Bolivia		United States		
Category	Item	UASB-Pond	3-Pond	Household	Community	City
System Boundary	Bathrooms	✓	✓	N/A	N/A	N/A
	Collection	✓	✓	N/A	✓	✓
	Treatment	✓	✓	✓	✓	✓
	Water Reuse	✓[a]	✓[a]	✓	✓	✓
	Agricultural water offsets	✓	✓	N/A	N/A	N/A
	Potable water offsets	N/A	N/A	✓	✓	✓
	Chemical fertilizer offsets	N/A	N/A	✓	✓	✓
	Energy offsets	✓	N/A	N/A	N/A	✓
Life Stages Considered	Construction	✓	✓	✓	✓	✓
	Operation and Maintenance (O&M)	✓	✓	✓	✓	✓

[a]Operation and maintenance phase only. Check mark means that the item is included in the system boundary or life stage.

2.5.1.2 Input Data and Life Cycle Inventory

Input data and emission sources considered in the LCA framework are similar to those developed in the WWEST model, but applicable to any LCA analysis tool (e.g., SimaPro 7.2, GaBi). These inputs include: (1) material production (i.e., material type, service life, purchase frequency, etc.), (2) material delivery (e.g., mode of transportation, distance traveled, material origin, etc.), (3) equipment operation (e.g., equipment type, use amount, use frequency), (4) energy production (e.g., electricity use, fuel use, energy recovery processes, etc.), (4) Direct emissions (Influent and effluent BOD, population served, etc.) and (5) sludge disposal (e.g., amount/year, facility type, gas recovery, transportation distance, etc.) based on Stokes and

Horvath (2010). These input parameters can be analyzed in process-based LCA models (SimaPro, Gabi) to evaluate the environmental impact of WWTPs with resource recovery.

Input data for the life cycle inventory (LCI) was compiled through data collection during site visits and correspondence with engineers and operators. Table 14 reviews the input data needed to calculate or obtain life cycle inventory data for the existing treatment systems. Similar to Stokes and Horvath (2006) and Stokes and Horvath (2010), the life cycle inventory compiled data on material production, material delivery, equipment operation, energy production, and sludge disposal.

Table 14. Model input data collected and inventory items for existing systems in current framework used for this investigation

Model Inputs	Inventory Items
Material Production: Material Type, Material Properties (kg, m^2, or m^3), Service Life (years), Purchase Frequency (qty)	Mass (kg), area (m^2) or volume (m^3) of materials (as required) used over 20-year lifespan
Material Delivery: Material Origin (City), Distance (km), Cargo Weight (tons), Mode of Transportation (vehicle type)	Freight transportation quantity (tkm) of materials delivered to sites over 20-year lifespan
Equipment Operation & Energy Production: Equipment Type, Power use (HP), Use Amount (hours), Use frequency, Fuel Type	Energy used (kWh) and fuel consumed (kg) by on-site equipment over 20-year lifespan
Sludge Disposal: influent TSS (mg/L), material production and delivery, equipment operation, and energy production input data (See above)	Fuel consumed (kg) by on-site equipment over 20-year lifespan (for sludge disposal)
Biogenic Emissions: Influent and effluent BOD_5 (mg/L) or COD (mg/L) data, influent TKN-N (mg/L) and influent flow rate.	CH_4 (kg), N_2O and CO_2 (kg) air emissions over 20-year lifespan

Note: Inputs were adapted from Stokes and Horvath (2006)

Biogenic greenhouse gas (GHG) emissions were considered, using equations developed by the EPA under Contract No. EP-D-06-118 (EPA, 2010). Biogenic emissions come from biological sources as opposed to fossil-based source (e.g., combustion of fossil fuels), which are generated through the combustion of fossil fuels. Few studies have considered biogenic sources from wastewater treatment process (e.g., biogenic methane from lagoons, nitrous oxide from nitrification), which can contribute significantly to the carbon footprint of natural systems and anaerobic treatment technologies (e.g., upflow anaerobic sludge blanket reactor (UASB)) when methane isn't flared or captured for beneficial reuse. It's important to include CH_4 and N_2O since their global warming potential is high at 25 and 298, respectively (IPCC, 2007). Biogenic CO_2 is considered to be carbon neutral by the Intergovernmental Panel on Climate Change (IPCC, 2006). Energy recovery, nutrient recycling and water reclamation at both sites were also considered. These resource recovery strategies were assessed as co-products providing a mitigation potential benefit. Enhancements to the WWEST framework included additional inputs and emissions sources for an integrated resource recovery framework. Additional input data collected for the life cycle inventory included:

- Total nitrogen (TN) and total phosphorus (TP) discharged to surface water from the treated effluent
- TN and TP discharged to soil via water reuse and associated fertilizer offsets
- TN and TP discharged to soil from biosolids land application and fertilizer offsets
- Chemicals and energy used to treat potable water (if available) and associated potable water offsets obtained through water reuse
- Energy offsets obtained through energy recovery

2.5.1.3 Environmental Impact Categories and Life Cycle Assessment

Sustainability indicators selected include: (1) carbon footprint (as global warming potential (GWP) in $kgCO_2eq$) using the Intergovernmental Panel on Climate Change (IPCC) 2007 GWP 20a; (2) embodied energy (as cumulative energy demand (CED) in MJ) quantified using the Cumulative Energy Demand methods (Hischier et al., 2010), and (3) eutrophication potential (EP as PO_4) using Eco-indicator 95 (Goedkoop, 1995).

The carbon footprint is defined as the sum of individual greenhouse gas emissions, in which carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are expressed in carbon dioxide equivalents (CO_2eq) by converting CH_4 and N_2O emissions using their global warming potential (IPCC 2006; Mihelcic et al. 2013). The carbon footprint includes direct process emissions (Scope 1), indirect emissions from on-site electricity consumption (Scope 2), and other indirect emissions from the production of materials, chemicals, fuels, etc. (Scope 3). Embodied energy includes both direct energy consumed on-site (e.g., electricity, fuel) and indirect energy from off-site sources (e.g., production of materials, chemicals). Eutrophication potential accounts for direct sources (i.e., N and P soil and water emissions from run-off and leaching) and indirect sources (i.e., NO_x air emissions deposited to aquatic environments) of anthropogenic eutrophying substances that lead to algal biomass formation in aquatic environments (Huijbregts and Seppala, 2001). A fate and transport method is embedded in LCA software to calculate aquatic eutrophication potentials to air, water, and soil, where aquatic environments are assumed to be N and P limited, leading to a conservative estimate of eutrophication potential (Refer to Huijbregts and Seppala (2001) for a detailed explanation on the fate analysis used to calculated aquatic eutrophication potentials).

Based on the analysis of varying frameworks, models and methods, a process-based LCA model (e.g., SimaPro 7.2 and SimaPro 8) was determined to be the most appropriate tool to assess the environmental impact of systems in both developing and developed world settings at different scales in this study. Various databases were used, which contain background data accounting for upstream processes (i.e., raw materials extraction, manufacturing, processing, etc.) (St. Gallen, Switzerland). Ecoinvent was the primary database used, but other databases were used when inventory items were not available. Other databases utilized include U.S. Life

Cycle Inventory LCI Database (USLCI), USA Input Output Database, LCA Food DK, and European Life Cycle Database (ELCD). Results for the entire system and each unit process were then interpreted to determine the embodied energy, carbon footprint, and eutrophication potential per functional unit.

2.5.1.4 Sensitivity Analysis and Uncertainty Analysis

A sensitivity or uncertainty analysis was conducted to determine how changes in inputs impact the results. To analyze the sensitivity of results, the input inventory values were modified by ±20% and output values were re-calculated. The difference between the +20% output and -20% output was divided by the original output and then divided by the percent change of the ±20% input terms divided by the original input. This calculates the sensitivity factor (SF), in which values closer to 1 are more sensitive and values closer to 0 are less sensitive.

A Monte-Carlo uncertainty analysis was used to assess the uncertainty of results in SimaPro 8 for the U.S. based systems. The Monte-Carlo method evaluates uncertainty by using random variables in the range of uncertainty to re-calculate results of each LCI input for 1,000 iterations (Pre et al., 2013). This method can subsequently be used to calculate an uncertainty distribution and provide insight on the uncertainty of the results.

2.5.2 Summary of Proposed Framework for WWTPs with Integrated Resource Recovery

After synthesizing data on system boundaries, phases considered, input data requirements, emission sources considered, major environmental impact categories relevant to resource recovery, and appropriate assessment methods, a framework was proposed to assess the life cycle impacts of WWTPs with integrated resource recovery. The proposed framework is a process-based LCA incorporating a comprehensive and consistent system boundary to make accurate comparisons of key environmental impact categories. In this research, carbon footprint and embodied energy represent global level impacts, whereas eutrophication potential represents local impacts to the water-energy-carbon-nutrient nexus. This framework could be applied to other countries and other settings to investigate wastewater treatment technologies and resource recovery strategies applicable to different scales and different contexts.

CHAPTER 3: SCALE'S INFLUENCE ON THE ENVIRONMENTAL SUSTAINABILITY OF WASTEWATER TREATMENT PLANTS WITH INTEGRATED RESOURCE RECOVERY

3.1 Introduction

Many wastewater treatment plants (WWTPs) worldwide require relatively high levels of energy (e.g., pumping, aeration) and resource consumption (e.g., materials, chemicals) to transport and treat wastewater (Muga and Mihelcic, 2008; CSS, 2009; Mo and Zhang, 2012; Mo and Zhang, 2013). In the United States, the water and wastewater industry is the third largest consumer of U.S. electricity, accounting for 3.4% of the total U.S. electricity consumption (EPRI, 2002; EPRI, 2009). Furthermore, in a typical U.S. city, up to 24% of energy usage by public utilities can come from wastewater treatment, though this varies regionally (Means, 2004; Mo et al., 2012). Population growth, climate change, rising water demand, aging infrastructure, and nutrient management place additional stressors on WWTPs to meet stringent discharge criteria while sustainably managing their energy consumption and associated carbon footprint over the life cycle (Zimmerman et al., 2008; Major, 2011; NAE, 2012; Padowski, 2013). Concerns over the sustainability of WWTPs have thus led to a paradigm shift in which wastewater is viewed as a renewable resource instead of simply a waste that must be treated to meet discharge standards (Guest et al., 2009).

Integrated resource recovery via water reuse, nutrient recycling, and energy recovery is beneficial to reducing the environmental impacts of WWTPs, highlighting the synergies of the water-energy-nutrient nexus (Mo and Zhang, 2013; Verbyla et al., 2013; Cornejo et al., 2014).

Water reuse can offset the energy and associated carbon footprint of potable water production (Friedrich et al., 2009; Meneses et al., 2010; Pasqualino et al., 2010; Stillwell and Webber, 2010; de Haas et al., 2011; Mo and Zhang, 2012). Additionally, treatment and water reuse can reduce the risk of eutrophication in local water bodies by reducing nutrient loads discharged directly to surface water (Hospido et al., 2004; Lundie et al., 2004; Meneses et al. 2010; Cornejo et al., 2013). Simultaneously, nutrients reclaimed from wastewater may be subject to runoff and groundwater infiltration that can lead to nutrient pollution problems if not properly applied and monitored. Despite this drawback, nutrient recycling can lead to the beneficial replacement of synthetic fertilizers, addressing phosphorus scarcity issues and improving the management of the nitrogen cycle (Lundie et al., 2004; Mihelcic et al., 2011; Fine and Hadas, 2012; NAE, 2012). In addition, energy recovery has been found to offset the energy and associated carbon footprint of WWTPs through the production of biogas via anaerobic digestion (Lundie et al., 2004; Mo and Zhang, 2012; Cornejo et al., 2013).

Another factor known to influence the sustainability of WWTPs is system scale or level of centralization. Most studies on this topic have investigated how scale influences the cost of WWTPs (EPA, 1980; Fraas and Munley, 1984; Fraquelli and Giandrone, 2003; Maurer et al., 2006) and WWTPs with resource recovery (Fane et al., 2002; Lee et al., 2013). However, local (e.g., nutrient management to protect local streams) and global (e.g., energy efficiency to reduce carbon footprint) concerns have led to a growing interest in scale's influence on the environmental sustainability of WWTPs (Tillman et a., 1998; Dennison et al., 1998; Lundin 2000; Pitterle, 2009; Shehabi et al., 2012). These studies used life cycle assessment (LCA) to investigate the impact of scale on the environmental sustainability of WWTPs with varying resource recovery applications in Sweden, United Kingdom, and the United States. Previous research varies in system boundary definitions, the number and types of environmental impact categories investigated, and the focus of study. For example, several European studies focus on scale's influence on WWTPs with source separation schemes (Tillman et a., 1998; Lundin 2000) and sludge management options (Dennison et al., 1998). These studies highlight the benefits to centralization due to economies of scale, and the benefits of nutrient recycling from urine to offset synthetic fertilizers. However, integrated resource recovery including water reuse, energy recovery, and nutrient recycling are not considered holistically in these studies.

Only two studies were identified that investigate the impact of scale on the environmental sustainability of WWTPs integrated with resource recovery in a United States context (Pitterle, 2009; Shehabi et al., 2012). Both of these studies found that centralized systems had a lower environmental impact due to economies of scale in terms of carbon footprint and life cycle energy consumption. Whereas both studies consider fertilizer offsets associated with nutrient recycling from biosolids, neither study considers fertilizer offsets associated with nutrients in reclaimed water. Pitterle (2009) considers offsets associated with river water replacement, but not potable water replacement. Other studies; however, have found that potable water offsets of embodied energy and carbon footprint from water reuse are greater than both energy recovery and nutrient recycling combined (Mo and Zhang, 2012). Furthermore, neither study considers how scale impacts eutrophication potential, which has been shown to be an important environmental impact category for wastewater management (Dennison et al., 1998; Hospido et al., 2004). The trade-offs between global impacts (e.g., carbon footprint and embodied energy) and local impacts (e.g., eutrophication potential from nutrients discharged to local water bodies) are important to consider, since they can lead to environmental problem shifting between carbon, energy, and nutrients (Foley et al., 2010).

Accordingly, this research uses three Florida case studies to evaluate how scale of implementation (i.e., household, community, city scale) impacts the environmental sustainability of WWTPs with integrated resource recovery. Embodied energy and carbon footprint are used in this study to investigate the global significance of impacts related to climate change and eutrophication potential to assess local impacts related to nutrients discharged to local water bodies. Eutrophication potential is of particular interest in many parts of the world (including Florida), where reducing nutrient pollution is crucial to ensuring ecosystem health and water quality protection.

3.2 United States Case Study Background

Tampa, FL was selected as the site location for this investigation because it is representative of major cities worldwide in nutrient sensitive coastal regions; facing trends of growing population, urbanization, and increased vulnerability to climate change impacts (Hallegatte et al., 2013). Furthermore, three systems representing proven household, community, and city scales of implementation were selected for analysis. Results are therefore transferable to other regions due to the selection of proven technologies used in common size ranges.

Proven WWTP technologies used for water reuse applications at the household (less than 0.1 mgd), community (0.1-1 mgd), and city (1-15 mgd) scale were investigated (See Table 15 and Figures 6, 7, and 8). Nationally, an estimated 26.1 million homes (20%) treat wastewater via septic systems at the household level, representing an important sector of wastewater management crucial to the permanent infrastructure of treatment in the United States (EPA, 2008). At the community scale, over 80% of wastewater treatment plants in U.S. are less than 5 mgd (Muga and Mihelcic, 2008) and 59% of the wastewater treatment systems with reuse in

Florida are implemented at a scale of 0.1-1 mgd (FDEP, 2014b). At the city scale, approximately 38% of Florida's reuse systems are implemented at a level of 1-15 mgd, where systems above 15 mgd (15-160 mgd) are far less common in Florida and nationwide (e.g., in Florida, 3% of the WWTPs with reuse are above 15 mgd) (Vedachalam and Riha, 2013; FDEP, 2014b). A critique of the Shehabi et al. (2012) investigation on the impact of scale is that the

Table 15. Systems investigated in Florida case studies

Scale	Population served	System	Treatment Processes	End-Use of Water
Household (250 gpd)	2-3	Septic Tank with aerobic treatment unit (ATU)	Primary tank, secondary (aerobic treatment unit)	Subsurface landscape drip irrigation
Community (0.31 mgd)	1,500	Advanced Water Reclamation Facility (WRF)	Headworks, aeration, denitrification tanks, re-aeration, clarification, de-nitrification filters, clearwell, chlorination, UV, aerobic digestion	Golf course irrigation and some surface water discharge
City (10.3 mgd)	100,000	Advanced WRF	Headworks, biological secondary treatment, clarification, filtration, chlorination, anaerobic digestion	Landscape irrigation and deep well injection

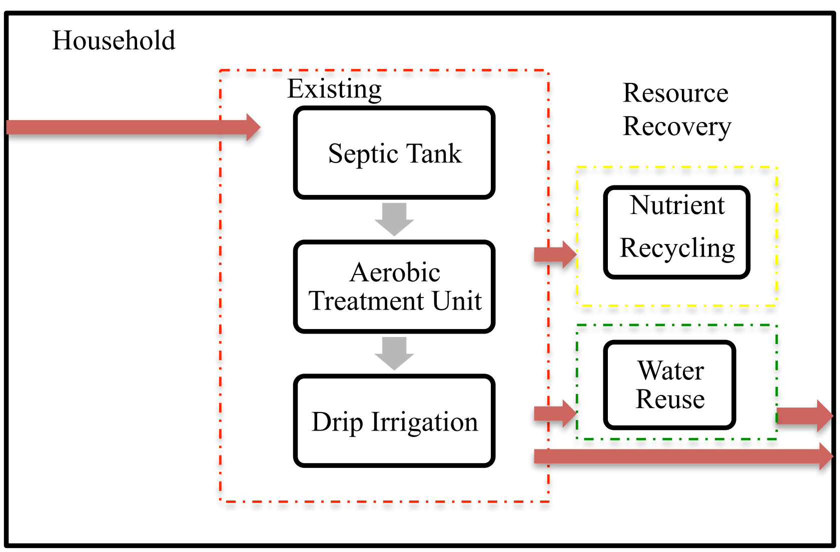

Figure 6. Process flow diagram of household system analyzed in U.S.

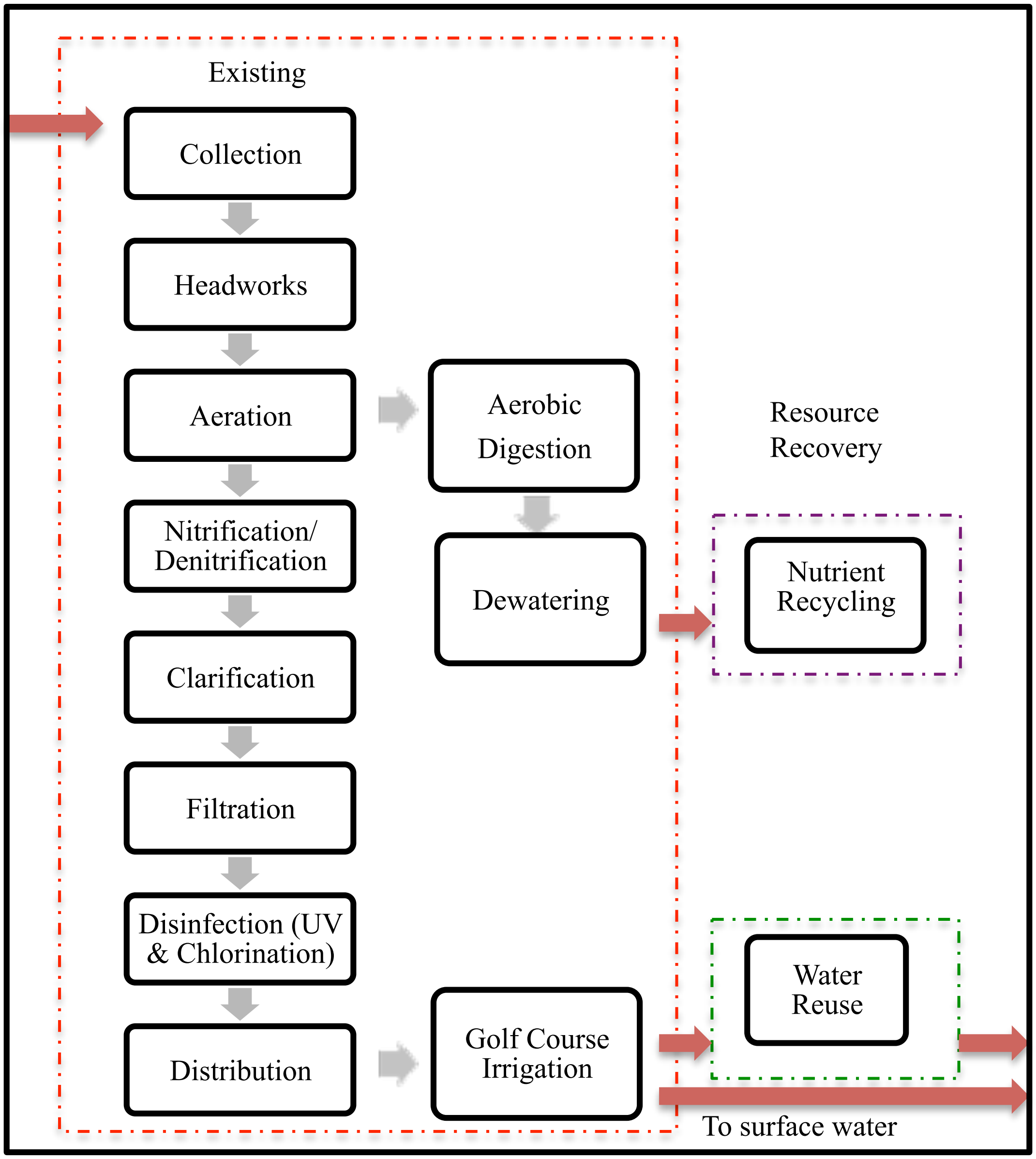

Figure 7. Process flow diagram of community system analyzed in U.S.

centralized system selected is too large (66.5 mgd) compared to other centralized plants (Vedachalam and Riha, 2013). Consequently, the systems selected in this study fall in the range of representative household, community, and city scale systems.

The three systems analyzed include: (1) a 250 gallon per day (gpd) septic tank followed by an aerobic treatment unit serving 1 home (2 to 3 people), and subsurface drip irrigation (2) a 0.31 million gallons per day (mgd) community water reclamation facility with nitrification and

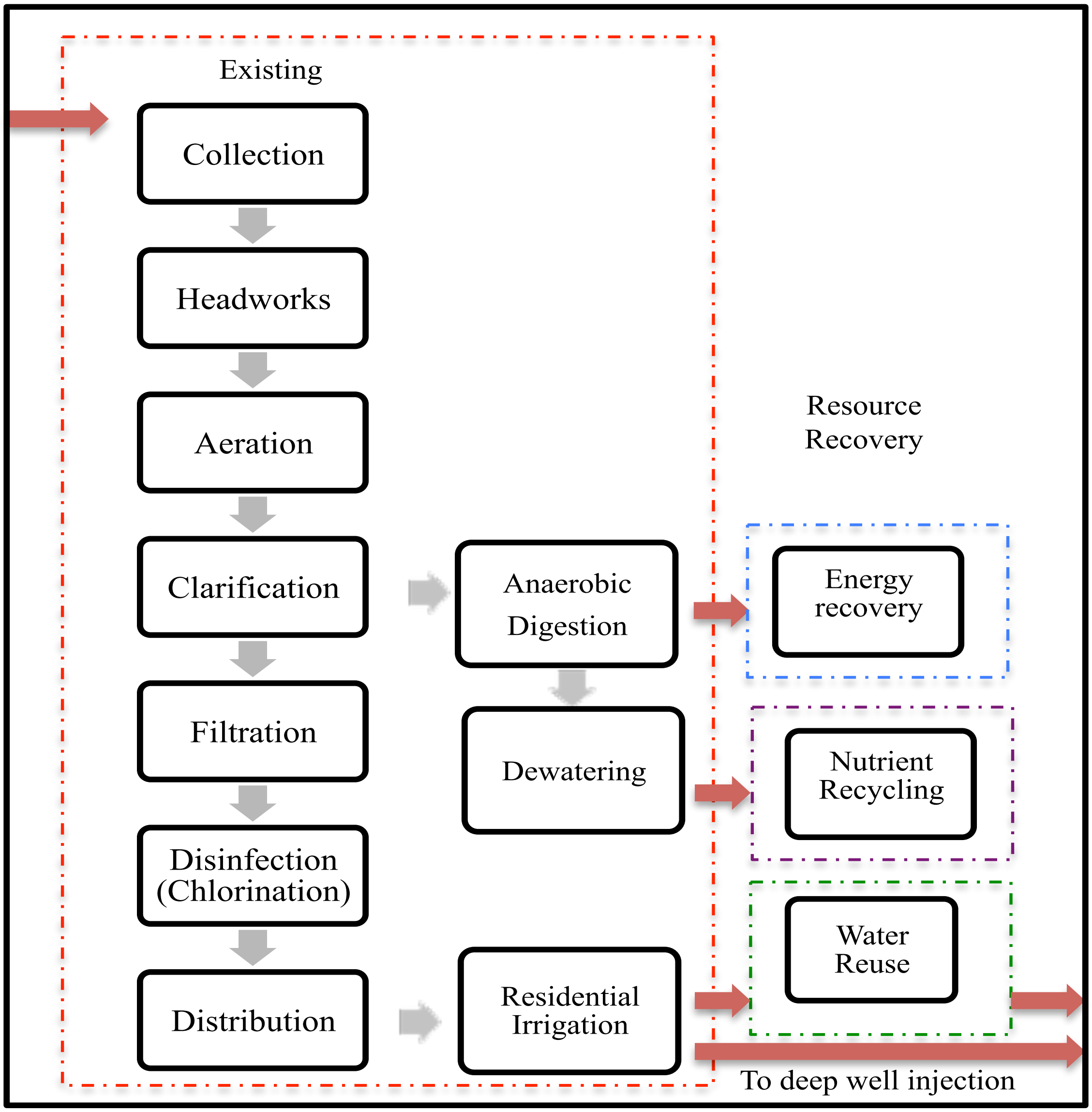

Figure 8. Process flow diagram of city system analyzed in U.S.

denitrification using headworks (grit removal, bar screens, odor scrubbing), equalization tanks, aeration tanks, denitrification tanks, re-aeration, clarifiers, denitrification filters, clearwell, chlorination and UV disinfection, aerobic digestion serving approximately 1,500 population equivalents (p.e.) with golf course irrigation reuse and some surface water discharge (3) a 10.3 mgd city scale advanced water reclamation facility with headworks (grit removal, bar screens), activated sludge (biological secondary treatment including aeration and return activated sludge), secondary clarification, filtration, chlorination, anaerobic digestion for energy recovery serving approximately 100,000 p.e. with residential landscape irrigation reuse and some deep well injection to prevent salt water intrusion. Operating parameters and key performance metrics are summarized in Table 16.

Table 16. Operating parameters and key performance metrics for U.S. systems

Parameter	Household	Community	City
Wastewater treatment standard	Secondary biological treatment for subsurface drip irrigation reuse	Advanced treatment with nitrogen removal for surface water discharge & reuse	Advanced biological treatment for reuse & deep well injection
BOD_5 in treated effluent(mg/L)	30 (20 - 40)[a]	1.8 (0.8 - 3.5)	2.1 (1.2 - 2.4)
Percentage of water reclaimed (%)	100	77	56
Effluent TN to soil from reclaimed water (mg/L)	16 (2 - 31)	0.23 (0.03-6.8)	2.3 (1.3-3.1)
Effluent TP to soil from reclaimed water (mg/L)	0.16 (0.12-0.20)	0.005 (0.004-0.04)	0.01 (0.004-0.03)
Total biosolids production (kg/yr):	9.8[a]	60,000	2,894,136

Note: Numeric values presented are average values, where values in parentheses are minimum and maximum values. [a]Asano et al. (2007)

Each scale implements different reuse and disposal methods. At the household, community, and city scale 100%, 77%, and 56% of the treated effluent is reclaimed. At the household level all the water can be reclaimed through subsurface drip irrigation, leading to reuse for residential irrigation and de-facto aquifer recharge. At the community scale nitrogen removal is practiced since around 23% of the water is discharged to surface water during the rainy season. At the city scale deep well injection is used to inject secondary treated effluent from WWTPs deep into the confined aquifer to provide aquifer recharge and dispose of wastewater. This practice is done in regions where other methods of disposal aren't feasible (FDEP, 2014c). Consequently, centralization of WWTPs may lead to a lower percentage of water reclamation and greater levels of aquifer recharge since larger systems produce greater volumes of wastewater that require disposal or reuse.

3.3 Methodology for United States Case Study

3.3.1 Goal and Scope Definition

Life cycle assessment (LCA) was used to evaluate the environmental impacts of WWTPs with integrated resource recovery at three different scales of implementation in Tampa, FL. Following ISO 14040 guidelines for LCA, the goal and scope were defined, a life cycle inventory was collected, a life cycle impact assessment was conducted, and results were subsequently interpreted (ISO, 2006). A functional unit of one cubic meter of treated water over a 20-year lifespan was selected. Previous LCAs have used lifespans between 20 and 50 years for wastewater infrastructures (Ortiz et al., 2007; Zhang et al., 2010; Lyons et al., 2009), where 20 years is a conservative lifespan for water infrastructures that typically last 20 to 50 years prior to major rehabilitation needs (EPA, 2012c). Infrastructure, operation and maintenance phases for collection, treatment, water reuse distribution, and integrated resource recovery stages were included in the system boundary. Water reuse and nutrient recycling occurs at all scales, whereas energy recovery only occurs at the city scale. Potable water offsets associated with reclaimed water, energy offsets associated with energy recovery, and fertilizer offsets associated with nutrients recovered were considered through system expansion.

3.3.2 Life Cycle Inventory

A life cycle inventory (LCI) of infrastructure (e.g., piping, tanks), energy (e.g. electricity, diesel), chemicals (e.g., coagulation/flocculation chemicals, disinfection chemicals), direct emissions (CH_4 and N_2O), nutrients emissions (e.g., nutrients discharged to surface water, nutrients discharged to soil via reclaimed water and biosolids), and resource recovery offsets (e.g., potable water, fertilizer, and energy offsets) was compiled (See Appendix A for comprehensive LCI of U.S. systems). A decentralized cost estimation tool (WERF, 2010) provided infrastructure and energy data for the household system, while plant operators and staff provided LCI data for infrastructure, energy, and chemicals at the community and city scale. A Florida energy mix (23.65% coal, 4.42% oil, 54.83% gas, 0.63% other fossil, 1.74% biomass, 0.01% hydro, 14% nuclear, 0.005% solar, 0.7% unknown/other purchased fuel) was used to calculate the carbon footprint impacts from electricity production (EPA, 2014b).

Methane (CH_4) emissions from anaerobic treatment processes and nitrous oxide (N_2O) emitted during treatment were calculated using an EPA method (Chandran, 2010; EPA, 2010), where details on calculations are shown in Appendix A. Biogenic CO_2 was not considered in accordance with IPCC guidelines (IPCC, 2006). Methane emissions are calculated based on inputs including wastewater influent flow rate, influent BOD_5, BOD_5 removal efficiency, and assumed constants (e.g., conversion factor for CH_4 generation, methane correction factor for specific wastewater treatment processes, fraction of carbon as CH_4, biomass yield of specific treatment processes) (EPA, 2010). Nitrous oxide emissions from wastewater are calculated based on a method recommended by Chandran (2010) that requires inputs related to the influent flow rate, influent TKN and assumed constants (e.g., N_2O emission factor, molecular weight conversion factor). Nitrous oxide (N_2O) emitted during land application of biosolids was estimated using an IPCC method that requires the annual amount of biosolids applied to soils and assumed constants (e.g., nitrogen additions from organic amendments) (IPCC, 2006). The methodology to calculate nitrous oxide from WWTPs accounts for variations in N_2O emissions from WWTPs using data collected from 12 WWTPs in the United States (Ahn et al., 2010). Consequently, this method more accurately estimates N_2O emissions from WWTPs compared to previous methods that use single emission factors related to protein intake and population (EPA, 2010). Despite these improvements, there is uncertainty associated with the methane and nitrous oxide calculations since measurements weren't taken directly on-site and assumed constants or input parameters may vary with site-specific conditions. Additionally, typical literature values for nutrient discharges were used (Asano et al., 2007) when data were not available, whereas nitrogen discharged to the soil via water reuse and biosolids was calculated assuming plant uptake of nitrogen ranging from 23% to 90% and plant uptake of phosphorus at 98% (Martinez and Clark, 2009). The difference between the total nutrients discharged and the plant uptake represent an emission to soil when reclaiming water and biosolids.

Water reuse offsets from local potable water production in Tampa, FL were calculated using LCI data from a previous study (Santana et al., 2014) and fertilizer offsets were estimated assuming optimal application rates for nitrogen and phosphorus uptake from water reuse and land application of biosolids. In these case studies, it is assumed that reclaimed water offsets potable water production; as opposed to other forms of conventional water production (e.g., groundwater from wells). Nutrients in the reclaimed water and biosolids are assumed to offset fertilizer usage representing a maximum fertilizer offset potential, despite variations in actual practice (i.e., residents with reclaimed water may not reduce fertilizer usage in practice). The remaining nutrients emitted after the assumed plant uptake potential are considered to be emissions to soil. Biosolids are land applied, as opposed to other forms of biosolids handling (e.g., incinerating or landfilling biosolids). This provides a nutrient benefit as a soil amendment that offsets fertilizers, but also leads to soil emissions that contribute to eutrophication.

The anaerobic digestion system at the city scale is currently undergoing a construction upgrade to implement temperature phased anaerobic digestion, a cleaning and compression system for digester gas, a receiving station for fats, oils, and grease (FOG), and a generator for digester gas (EPA, 2015b). However, these items were excluded from the system boundary since they are still under construction. Consequently, the existing condition assumes that biogas is flared to reduce the impact of methane (CH_4) emissions and the energy recovery condition assumes that natural gas is avoided through biogas recovery from anaerobic digestion. Energy recovery from household and community systems in United States is not considered. Though previous research found that biogas recovery from household anaerobic treatment processes could lead to net energy production (Galvin, 2013), another study found that small-scale digesters have a high failure rate due to improper maintenance of biogas digesters (Bruun et al., 2014). Household biogas digesters and energy recovery from community systems less than 5 mgd (e.g., CHP at this scale is not cost-effective) are not common in developed world settings (EPA, 2007; Mo and Zhang, 2013) and are consequently not considered.

3.3.3 Life Cycle Assessment and Interpretation

The impact assessment was conducted in SimaPro 8 (PhD version) and background information from databases embedded in life cycle assessment (LCA) software were used to account for upstream impacts associated with the production of inventory items (PReConsultants, 2014). SimaPro 8 was utilized to analyze the impact of key impact categories including embodied energy (Cumulative Energy Demand method expressed as MJ/m^3), carbon footprint (IPCC 2007 GWP 100a method expressed as kg CO_2eq/m^3) and eutrophication potential (Eco-indicator 95 method expressed as g PO_4eq/m^3) (Goedkoop, 1995; Frischknecht et al., 2007; Hischier et al., 2010). It is important to note that carbon footprint results are expressed in kg CO_2eq/m^3, but CO_2, CH_4, and N_2O are included. Similarly, eutrophication potential is expressed as g PO_4eq/m^3, yet both nitrogen and phosphorus are included. A fate and transport model embedded in the eutrophication potential method is used to calculate impacts of nutrient discharges to the environment. Subsequent interpretation of the results identified dominant contributors to the selected impact categories to assess the impact of scale.

3.3.4 Uncertainty Analysis

Lastly, a Monte-Carlo uncertainty analysis was conducted using SimaPro 8 to evaluate the uncertainty associated with the LCA results. The Monte-Carlo method evaluates uncertainty by re-calculating the results for random variables within the uncertainty range of each LCI input for 1,000 iterations (Pre et al., 2013). This is then used to determine the distribution and provide insight on the uncertainty of the results.

3.4 Results and Discussion for United States Case Study

3.4.1 Impact of Scale on Embodied Energy

The total embodied energy of WWTPs decreases as scale of implementation increases from household (40.0±0.4 MJ/m^3) to community (33.8±1.0 MJ/m^3) to city scale (16.0±4.8 MJ/m^3) as shown on Figure 9. Whereas the city scale system falls in the range of embodied energy for water reuse systems (13-18 MJ/m^3) from previous studies (Stokes and Horvath, 2009; Pasqualino et al., 2010), the community and city scale systems have a high embodied energy compared to these studies.

In terms of collection, the embodied energy associated with wastewater collection increases with increased centralization. At the household scale, wastewater collection is assumed to be negligible due to limited piping and collection by gravity. However, the embodied energy of collection including both direct and indirect energy increases from community (1.4 MJ/m^3) to city scale (2.3 MJ, m^3). Additionally, the percent contribution from collection (i.e., collection piping and electricity) increases from 4% of the total embodied energy at the community scale to 14.7% at the city scale (Refer to Table 17). This finding coincides with a known trend, where energy and infrastructure costs for collection increase with centralization, since transport distance is higher and larger pipe diameters are needed for larger systems (Asano et al., 2007; EPA, 2013a).

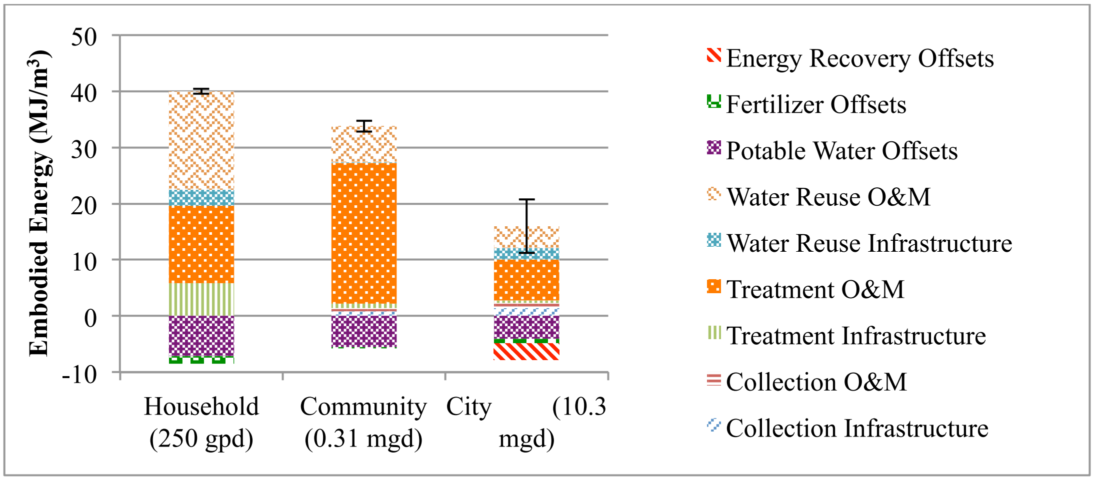

Figure 9. Embodied energy of wastewater systems with collection, treatment, water reuse, and resource recovery offsets at different scales

Table 17. Percent contribution from sewer collection, treatment, water reuse to the total embodied energy of wastewater systems and resource recovery offsets at different scales

Phase	Stage	Description	Household (250 gpd)	Community (0.31 mgd)	City (10.3 mgd)
Construction	Collection	Piping	-	2.5%	9.2%
	Treatment	Tanks	14.6%	2.3%	3.1%
	Reuse	Piping	7.2%	0.4%	12.5%
O&M	Collection	Electricity	-	1.5%	5.5%
	Treatment	Sludge Removal	0.02%	0.02%	0.04%
		Chemicals	-	6.3%	17.5%
		Electricity	34.3%	62.8%	22.5%
		Diesel	-	2.4%	5.5%
	Reuse	Electricity	43.8%	17.9%	15.4%
		Diesel	-	3.9%	8.9%
	Resource Recovery Offsets	Potable Water Offsets	-17.9%	-15.4%	-25.2%
		Fertilizer Offsets	-3.3%	-0.5%	-5.1%
		Energy Offsets	-	-	-18.5%

The treatment stage is a major contributor to the total embodied energy at the household (49%), community (74%), and city (49%) scales. For the treatment life stage, changes in treatment technology and associated electricity demand have a larger impact on the embodied energy than changes in scale of implementation. Treatment technology at the household level includes a septic tank and aerobic treatment unit (1.1 kWh/m^3), whereas treatment technology at the community scale includes nitrification/denitrification basins, UV, chlorination, and aerobic digestion (1.8 kWh/m^3). Whereas, community and city scale systems utilize similar primary and secondary treatment, they have different disinfection technologies, solids treatment technologies, and no nitrogen removal. Chlorination and anaerobic digestion used at city scale are less energy intensive compared with nitrogen removal, UV, chlorination and aerobic digestion used at community scale, leading to a decrease in treatment electricity consumption at the city scale (0.3 kWh/m^3). This leads to an increase in embodied energy of treatment from household (19.5 MJ/m^3) to community scale (25.7 MJ/m^3) and a decrease in embodied energy of treatment from community to city scale (7.8 MJ/m^3).

The results show that from the community to city scale, there is a decrease in embodied energy of treatment due to economies of scale and changes in treatment technology. This finding is consistent with previous LCA studies (Tillman et al., 1998; Lundin et al., 2000; Shehabi et al., 2013) that found that larger system benefit from lower energy intensities due to economies of scale. Furthermore, the low embodied energy at the city scale system (10.3 mgd) serving 100,000 p.e coincides with a study on 103 WWTPs in Italy that found that 100,000 inhabitants was the minimum efficient plant size for operational cost per cubic meter (Fraquelli and Giandrone, 2003). In that study, systems serving populations larger than 100,000 inhabitants do not benefit significantly from economies of scale in terms of cost.

The embodied energy associated with water reuse decreases with increased level of centralization, primarily due to a decrease in pumping energy required to distribute reclaimed water to end users as systems become larger. This finding differs from a commonly mentioned driver towards decentralization, in which energy needed for water delivery decreases as systems become smaller and pumping distances for water reuse delivery decrease (Shehabi et al., 2012; Lee et al., 2013). In the current study, electricity demand per cubic meter for household drip irrigation (1.4 kWh/m^3) is three times greater than electricity demand for water reuse distribution for golf course irrigation at the community scale and seven times greater than the electricity demand for water reuse distribution for residential irrigation at the city scale. This finding is counterintuitive, because drip irrigation is an efficient form of irrigation with a lower energy cost than spray irrigation (WERF, 2010). Perhaps, drip irrigation energy is larger because it's based on an energy estimate from the WERF decentralized cost tool, whereas community and city scale values come from actual electricity bills. Another reason why pumping energy per cubic meter of water treated decreases as systems increase in size, may be because larger pumps can be more energy efficient than smaller pumps (Satterfield, 2013). Additionally, the city scale system has a lower energy demand for water reuse pumping because variable frequency drive (VFD) pumps are used at this scale, as opposed to conventional pumps used at the community and household scales. This result shows that the benefits of energy efficient VFD pumps implemented at centralized treatment plants can outweigh the drawbacks of pumping reclaimed water to end-users at further distances under certain topographical conditions (e.g., Florida's flat topography).

Whereas previous studies in California have found conveyance energy costs to be 20-39.5 times higher than treatment costs (Cohen et al., 2004; Wolff et al., 2004; Guo et al., 2013); this Florida case study finds that treatment energy is only 0.8-3.5 times greater than conveyance for all systems. This suggests that water reuse has a lower or comparable energy demand than treatment for these particular systems in flat topography locations. Horizontal pumping consumes much less energy per cubic meter when pumping long distances, particularly when the velocity of horizontal pumping is kept low (S. Oakely, personal communication, March 24, 2015). This occurs because the energy to overcome total dynamic head (e.g., elevation) is greater than the energy to overcome minor friction losses (e.g., distance). For example, the energy cost of 100 km of horizontal pumping is equivalent to 100 m vertical pumping at \$0.05-0.06/m^3, highlighting that transporting water horizontally can be significantly less energy-intensive than pumping vertically (Zhou and Tol, 2013). Thus, flat topography locations may favor centralized wastewater management for water reuse for systems ranging from 250 gpd to 10.3 mgd, investigated in this research.

3.4.2 Impact of Scale on Embodied Energy Offset Potential of Resource Recovery

Water reuse is the most effective form of resource recovery, leading to the greatest energy offset potential at all scales (15-25%). The decentralized household scale benefits from greatest potable water offsets and these offsets decrease with system size, since the percentage of water reclaimed decreases as scale increases (e.g., water reclaimed is 100% at household, 77% at community, and 56% at city level). Mo et al. (2012) also found water reuse to be more beneficial than energy recovery and nutrient recycling at a 54.2 mgd WWTP in Tampa; however, another study on a 130 mgd WWTP in Denver found that energy recovery (30.6%) had a higher offset potential than water reuse and nutrient recycling (Pitterle, 2009). This highlights that energy recovery may be more significant for larger systems, whereas water reuse is more important for smaller systems. This finding coincides with another study, in which recovering water and nutrients was found to be more important than energy recovery for community scale systems serving around 1,000 p.e in rural Bolivia (Verbyla et al., 2013).

Fertilizer offsets associated with nutrient recycling are the least significant form of resource recovery, contributing to only a 0.5-5% offset of the total embodied energy at all scales.

Fertilizer offsets of embodied energy are low because the nutrient discharge load available to replace synthetic fertilizers is low. These offsets decrease from household to community scale and increase from community to city scale. At the household scale fertilizer offsets of embodied energy are the highest (1.3 MJ/m^3 energy offset), because nitrogen levels in the reclaimed water used for beneficial irrigation and available for fertilizer offsets are the highest (30.0 mg/L TN, 8.0 mg/L TP). At the community scale, nutrient recycling from reclaimed water have a lower mitigation potential (0.2 MJ/m^3 fertilizer offsets of embodied energy) because the reclaimed water has a lower nutrient content (0.001 mg/L TN, 0.0002 mg/L TP). As scale increases from community to city scale, fertilizer offsets increase (0.8 MJ/m^3 offset) due to higher nutrient level in reclaimed water available for offsets (0.009 mg/L TN, 0.006 mg/L TP). Additionally, nutrients associated with biosolids lead to increasing fertilizer offsets as systems become more centralized, primarily because the average concentration of nitrogen in biosolids increases from household (0.65 mg/L TN) to community (3.0 mg/L TN) to city scale (10.4 mg/L TN). Fertilizer offsets of embodied energy from phosphorus-based fertilizers avoided through biosolids land application and water reuse are less significant than offsets associated with nitrogenous fertilizer. Additionally fertilizer offsets from water reuse have a higher embodied energy offset potential than biosolids land application for the household scale, whereas biosolids have a slightly higher offset potential than water reuse at the community and city scale. This is likely due to the increased production of biosolids as systems become larger and more centralized. The increased production of biosolids also depends on treatment technologies implemented for solids handling.

Integrating resource recovery strategies was found to decrease the total embodied energy at all scales. At the city scale combining water reuse and energy recovery leads to clear advantages for the more centralized system, whereas energy recovery is not applicable at smaller scales. Energy recovery occurs at the larger city scale due to the implementation of anaerobic digestion. This leads to an energy offset of approximately 18%, where potable water offsets were approximately 25% and fertilizer offsets were 5% at the city scale. Consequently, the city scale provides the highest total percent offset potential, where integrated resource recovery offsets 49% of the total embodied energy. This is similar to Mo et al. (2012) findings of integrated resource recovery mitigating up to 61% of the total embodied energy at a 54.2 mgd WWTP facility in Tampa. In this study, integrated resource recovery at the city scale leads to a total offset potential of 7.8 MJ/m^3, which is approximately equal to the embodied energy need during the treatment stage and greater than the embodied energy needed to implement water reuse.

3.4.3 Impact of Scale on Carbon Footprint

The carbon footprint decreases as scale increases from household (3.3±0.3 kg CO_2eq/m^3) to community (2.1±0.1 kg CO_2eq/m^3) to city (1.1±0.2 kg CO_2eq/m^3) level for the selected wastewater treatment systems with integrated resource recovery (See Figure 10). This overall trend is similar to the total embodied energy because indirect (Scope 2) emissions associated with electricity are a dominant contributor (38-82%) at all scales (See Table 18). Direct emissions (Scope 1) associated with methane (CH_4) and nitrous oxide (N_2O) have a comparatively lower contribution ranging from 5-17% at all scales.

Direct emissions (Scope 1) decrease as scale increases despite fluctuations in methane (CH_4) and nitrous oxide (N_2O). Methane contributions are higher at the household (11%) level compared to the community level (negligible) due to changes in technology. Anaerobic treatment from the septic tank at the household level has the highest contribution to CH_4 emissions (0.36 kg CO_2eq/m^3), whereas community and city scale methane emissions are negligible. Similarly, Pitterle (2009) also found that CH_4 contributions were higher for septic systems compared to larger WWTPs, ranging from 34-42% of the total emissions. Household biogas digesters could be used to offset the carbon footprint of household systems; however, this

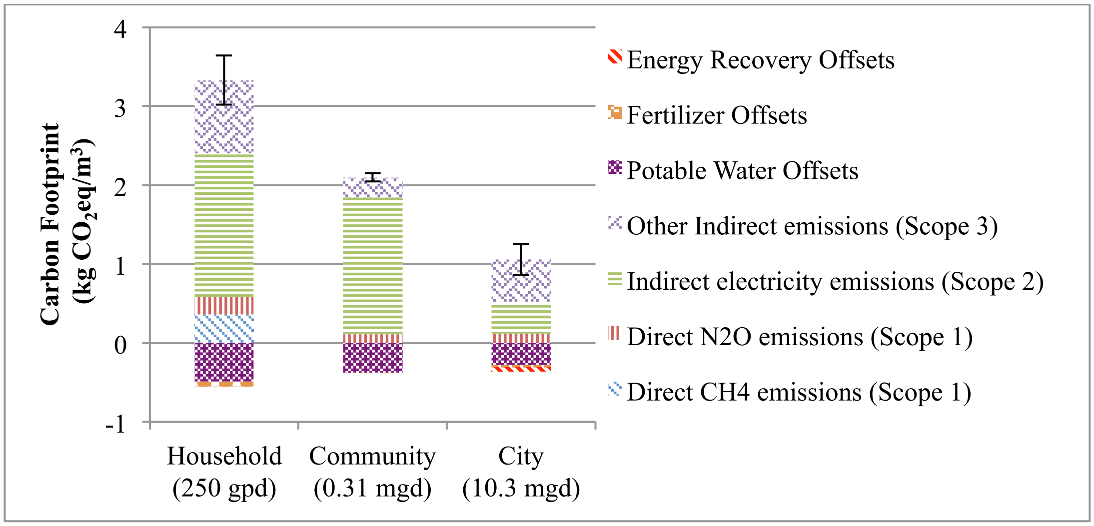

Figure 10. Carbon footprint of WWTP including scope 1, 2, and 3 emissions and resource recovery offsets at different scales

Table 18. Percent carbon footprint of WWTP including scope 1, 2, and 3 emissions and resource recovery offsets at different scales

Phase	Stage	Item	Household (250 gpd)	Community (0.31 mgd)	City (10.3 mgd)
Infrastructure	Sewer	Piping[a]	-	1.4%	8.3%
	Treatment	Tanks[a]	19.6%	4.6%	5.5%
	Reuse	Piping[a]	8.2%	0.2%	15.7%
O&M	Sewer	Electricity[b]	-	1.5%	4.8%
	Treatment	Chemicals[a]	-	5.9%	18.7%
		Direct CH_4[c]	10.9%	0.0%	0.0%
		Direct N_2O[c]	6.3%	4.4%	6.7%
		Electricity[b]	24.1%	62.4%	19.8%
		Diesel[b]	-	0.4%	0.9%
		Sludge Removal[a]	0.01%	0.02%	0.04%
		Direct N_2O (biosolids)[c]	0.1%	0.7%	4.6%
	Reuse	Electricity[b]	30.8%	17.7%	13.6%
		Diesel[b]	-	0.7%	1.4%
	Resource Recovery Offsets	Potable Water Offsets	-14.7%	-17.8%	-25.9%
		Fertilizer Offsets	-1.9%	-0.4%	-3.7%
		Energy Offsets	-	-	-3.9%

[a]Scope 3 emissions. [b]Scope 2 emissions. [c]Scope 1 emissions

technology may be more prone to failure (Bruun et al., 2014) and entails greater operational training requirements. At the community scale, CH_4 contributions are negligible due to use of aerobic treatment processes for BOD removal, nitrogen removal and aerobic digestion; however, the aeration requires additional electricity, highlighting a tradeoff between aerobic and anaerobic treatment processes for biosolids at WWTPs. At the city level, CH_4 emissions are also negligible when flared or recovered from the anaerobic digester, but can contribute to the carbon footprint when emitted directly (0.11 kg CO_2eq/m^3).

Nitrous oxide (N_2O) emissions decrease from household (0.21 kg CO_2eq/m^3) to community scale (0.11 kg CO_2eq/m^3), primarily because the influent total nitrogen load decreases with increased scale for these particular systems. Nitrous oxide (N_2O) emissions from community and city scale (0.12 kg CO_2eq/m^3) are comparable, because contributions of N_2O from land applied biosolids increase with scale from household (0.1%) to community (0.7%) to city (4.6%) scale. This is due to the rise in concentration of nitrogen present in biosolids as the level of centralization increases. It is important to note that previous LCA studies on the influence of scale have largely ignored N_2O emissions, despite the high global warming potential of nitrous oxide, 298 times more potent than CO_2 (IPCC, 2007). The percent contribution from direct N_2O emissions during treatment at all scales (4.4-6.7%) is slightly higher than previous estimates of nitrous oxide's contribution (3%) to the total carbon footprint of wastewater systems (EPA, 2009; Ahn et al., 2010). Consequently, previous research suggests that minimizing ammonium or nitrite build up in activated sludge processes could lead to lower N_2O emissions, particularly when dissolved oxygen is present (Ahn et al., 2010). Ahn et al. (2010) suggests that this can be achieved by decreasing over-aeration, which has the additional benefit of reduced electricity consumption and avoiding incomplete or discontinuous nitrification.

Similar to operational energy, Scope 2 emissions associated with electricity production decrease with increasing level of centralization. Indirect (Scope 2) emissions follow the same trend as electricity consumption. Scope 2 emissions are dominant contributors at the household (55%), community (82%), and city (38%) level. This trend was found despite fluctuations in treatment electricity, where changes in technology can be more important than changes in scale. Scope 2 emissions associated with treatment electricity initially increase from household (0.80 kg CO_2eq/m^3) to community scale (1.3 kg CO_2eq/m^3), and then decrease from community to city scale (0.21 kg CO_2eq/m^3). Scope 2 emissions associated with distribution electricity for water reuse represent 31%, 18%, and 14% of the relative carbon footprint at the household, community, and city scale respectively. These emissions decrease with increasing scale, where this trend is likely attributed to more efficient VFD pumps used at larger scales.

Overall Scope 3 indirect emissions associated with material and chemical production initially decrease and then increase as scale goes from household (0.92 kg CO_2eq/m^3, 28% of total) to community (0.26 kg CO_2eq/m^3, 12% of total) to city scale (0.53 kg CO_2eq/m^3, 48% of total). The dominant contributor to scope 3 emissions at the household scale is treatment tanks, contributing to 20% of the total carbon footprint. This is consistent with previous studies of varying scales that find the environmental impact of infrastructure is larger for more decentralized systems (Tillman et al., 1998; Lundin et al., 2000; Pitterle, 2009; Shehabi et al., 2012). Carbon footprint of treatment tanks decrease as scale increases contributing to less of the carbon footprint at the community and city scale (5-6%). At the community scale, chemicals (6% of total) and treatment tanks (5% of total) are the largest scope 3 contributors. In contrast, at the city scale chemicals (19% of total) and water reuse piping (16% of total) are the largest scope 3 contributors. Similar to embodied energy, the carbon footprint of chemicals increases with scale since more chemicals are required to treat greater volumes of water. This finding is consistent with a previous study (Lundin et al., 2000) in which chemical usage for a large scale WWTP (72,000 p.e.) was higher than a small scale WWTP (200 p.e.).

Despite these increases, the overall carbon footprint (Scope 1, 2, and 3) at the city scale is still less than the community and household scale because Scope 2 emissions associated with electricity are dominant. Although the carbon footprint of community and city scale technologies fall into the range of carbon footprint of WWTPs integrated with resource recovery from previous studies (0.1 - 2.4 kg CO_2eq/m^3), the carbon footprint of the household system is higher than the range of emissions from previous studies (Mihelcic et al., 2013; Cornejo et al., 2014). This is likely due to the inclusion of aerobic treatment units and drip irrigation for reuse. Aerobic treatment units are beneficial because they improve the treatment of septic systems, thereby addressing the national and local concerns about failing septic systems (Gorman and Halvorsen, 2006; Halvorsen and Gorman, 2006; Cake et al., 2013). In addition, drip irrigation is a beneficial dispersal method designed for efficient water reuse and nutrient uptake by plants in the root zone near the soil surface (WERF, 2010). However, less-energy intensive aeration or passive techniques for nutrient reduction (Anderson et al., 1998; Hirst et al., 2014; Anderson et al., 2014) and gravity trenches designed to maximize reuse may be more beneficial to for energy-efficiency at this scale.

3.4.4 Impact of Scale on Carbon Footprint Offset Potential of Resource Recovery

Potable water offsets from water reuse are the dominant resource recovery strategy for carbon footprint mitigation. Water reuse can offset 0.5 kg CO_2eq/m^3 at the household scale, 0.4 kg CO_2eq/m^3 at the community scale and 0.3 kg CO_2eq/m^3 at the city scale by avoiding energy used to produce potable water. This represents relative carbon footprint offsets of 15%, 18%, and 26% for household, community, and city scale systems, respectively. This mitigation potential is lower than the carbon footprint offset potential of water reuse from a larger 54.2 mgd facility (e.g., 36-40% of the total carbon footprint) in another study (Mo and Zhang, 2012) because at larger scales, more water can be reclaimed and therefore more potable water can be replaced.

Consequently, water reuse not only leads to the greatest offsets of global impacts (e.g., carbon footprint) among the three resource recovery strategies, but can also lead to beneficial water savings in regions seeking to reduce potable water consumption for non-potable uses. This is important for arid areas like California that mandatory water restrictions for outdoor residential irrigation was recently put in place as a response to extreme drought conditions (Nagourney, 2015). In United States, a typical household consumes 320 gallons of water per day, where 30% is used for outdoor uses (e.g., watering lawns) (EPA, 2015c) and non-potable outdoor water usage increases in arid locations. In Florida, outdoor water usage can reach up to 50% of the household water usage (SWFWMD, 2015), highlighting the importance of water reuse. Replacing potable water with reclaimed water can therefore lead to carbon footprint reductions, while saving fresh water and reducing costs associated with potable water production.

Carbon footprint offsets through integrated resource recovery at the household (0.55 kg CO_2eq/m^3), community (0.39 kg CO_2eq/m^3) and city (0.36 kg CO_2eq/m^3) scales provide the greatest benefit, since resource recovery strategies are combined. Similar to embodied energy, fertilizer offsets of carbon footprint associated with nutrient recycling are less significant accounting for a 0.4-4% at all scales, whereas the city scale energy recovery leads to a 4% decrease in carbon footprint compared to flaring (e.g., methane gas is burned and most of it is converted to CO_2), which was conducted prior to the anaerobic digestion system undergoing construction. The integration of all possible resource recovery offsets account for 17% of the total carbon footprint at the household scale, 18% at the community scale, and 34% at the city scale. Consequently, integrated resource recovery effectively offsets scope 1 direct emissions at all scales.

3.4.5 Impact of Scale on Eutrophication Potential

Eutrophication potential accounts for nitrogen and phosphorus emissions to surface and ground waters that lead to algal blooms and is expressed as g of PO_4eq/m^3. Eutrophication potential decreases with scale from household (10.5±4.3 g PO_4eq/m^3) to community (3.6±1.1 g PO_4eq/m^3), and slightly increases with scale from community to city (4.4±1.5 g PO_4eq/m^3) level of implementation (Figure 11). This is largely due to shifts in treatment level and nutrient discharges as scale changes. For example, eutrophication potential from indirect O&M and infrastructure sources (e.g., piping, tanks, electricity, sludge removal, chemicals, diesel) contributes to 28% at the household level, 59% at the community level, and 27% at the city level (Table 19).

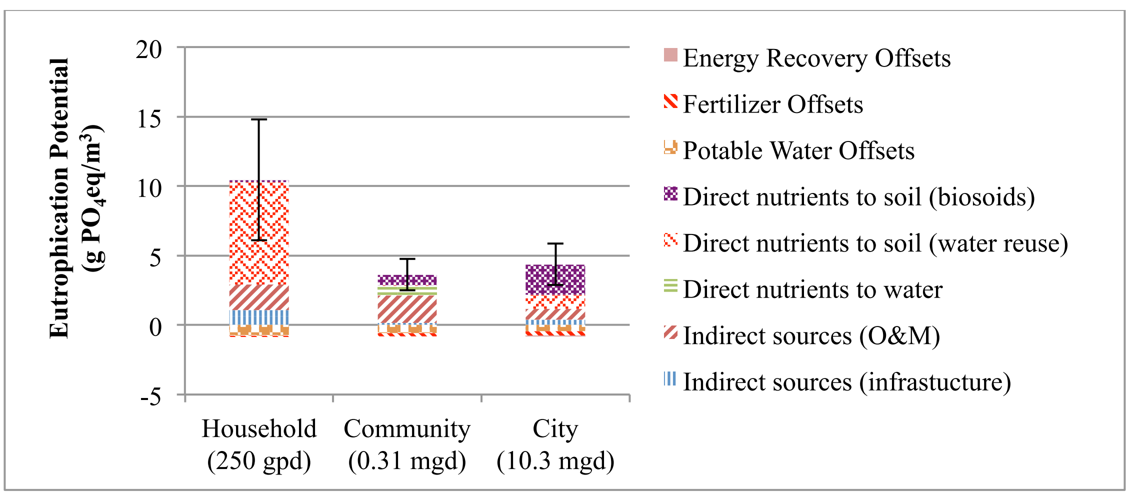

Figure 11. Eutrophication potential of systems including direct nutrients to soil and water, indirect sources of eutrophication, and resource recovery offsets at different scales

Table 19. Percent eutrophication potential of systems including direct nutrients to soil and water, indirect sources of eutrophication, and resource recover offsets at different scales

Phase	Stage	Item	Household (250 gpd)	Community (0.31 mgd)	City (10.3 mgd)
Infrastructure	Collection	Piping	-	0.4%	1.2%
	Treatment	Tanks	7.7%	3.3%	1.6%
	Distribution	Piping	2.6%	0.1%	5.3%
O&M	Collection	Electricity	-	0.9%	1.2%
		Sludge Removal	0.0%	0.0%	0.0%
	Treatment	Chemicals	-	6.4%	9.2%
		Electricity	7.8%	36.8%	4.9%
		Diesel	-	0.2%	0.1%
	Distribution	Electricity	9.9%	10.5%	3.3%
		Diesel	-	0.3%	0.2%
	Discharge	N to surface water	-	7.5%	-
		P to surface water	-	11.0%	-
		N to soil (water reuse)	65.7%	2.0%	22.2%
		P to soil (water reuse)	4.7%	0.3%	0.8%
		N to soil (biosolids)	1.2%	14.8%	43.3%
		P to soil (biosolids)	0.4%	5.4%	6.5%
	Resource Recovery	Potable Water Offsets	-7.1%	-15.9%	-9.6%
		Fertilizer Offsets	-0.9%	-6.3%	-8.2%
		Energy Recovery Offsets	-	-	-0.4%

For direct sources (e.g., nutrients discharged directly to the environment), eutrophication potential decreases from household (7.5 g PO_4eq/m^3) to community (1.5 g PO_4eq/m^3) scale and subsequently increases from community to city (3.2 g PO_4eq/m^3) scale. This trend can be largely attributed to changes in concentrated nitrogen loads discharged to the environment. The household system has the highest contribution from direct sources due to the high levels on nitrogen discharged to soil through water reuse (TN=16.4 mg/L, TP=0.16 mg/L), accounting for 66% of the eutrophication potential. The community scale system achieves the lowest eutrophication potential due to higher removal of nutrients (e.g., TN=0.23 mg/L, TP=0.005 mg/L in reclaimed water). Consequently, the community scale system has lower direct impacts than household and city scale systems, despite having direct eutrophication potential impacts from surface water discharge (8% from TN, 11% from TP), reclaimed water (2% from TN, 0.3% from

TP) and biosolids (15% from TN, 5% from TP). The dominant contributor from direct sources for the city scale is nitrogen emission to soil from biosolids (43%), followed by nitrogen emissions to soil from water reuse (22%).

In this study, the eutrophication potential associated with nitrogen discharged to soil from reclaimed water and biosolids is more significant than the eutrophication potential associated with phosphorus discharged to soils from reclaimed water and biosolids at all scales. It's important to note; however, that region-specific fate factors of air and soil (e.g., climate, plant uptake, land use, soil type) and limiting nutrients are not considered in the calculation of eutrophication potential used in SimaPro (Huijbregts and Seppala, 2001). The fate and transport model of aquatic eutrophication used in SimaPro assumes nitrogen (N) and phosphorus (P) are both limiting nutrients, leading to conservative estimates of eutrophication potential (Huijbregts and Seppala, 2001). In general, eutrophication potentials are based on the average chemical composition of aquatic organisms representing algae, $C_{106}H_{263}O_{110}N_{16}P$ accounting for the contribution of each of nutrients (primarily N and P) to biomass formation. One mole of biomass requires 16 moles of N and 1 mole of P. Therefore if the contribution of eutrophication of one mole of P is 1 and the contribution of one mole of N is 1/16, where PO_4 is as reference compound for eutrophication potential. The contribution of one mole is then expressed as the contribution of one gram by dividing by the molecular weight, where the reference substance is used to create eutrophication potentials. Nitrogen and phosphorus are treated separately in SimaPro's eutrophication potential method, where the final results depend on both characterization factor and the amount of nutrients released.

Direct nitrogen emissions from land applied biosolids increase with scale, where nitrogen in biosolids contribute to 1.2%, 15%, and 43% of the eutrophication potential at the household, community, and city scale. This increase in eutrophication potential from biosolids as scale increases is primarily due to an increase in the nitrogen load of biosolids from household (TN=0.3mg/L) to community (TN=1.3mg/L) to city level (TN=4.5 mg/L). Eutrophication potential associated with phosphorus discharged from biosolids is less significant, accounting for 0.4%, 5.4%, and 6.5% of the eutrophication potential at the household, community, and city scale, respectively. Whereas previous studies examining scale's influence on the life cycle impacts of WWTPs with integrated resource recovery have generally ignored eutrophication potential, these findings suggest that scale of implementation and level of treatment have an impact on eutrophication potential.

The community system has the lowest eutrophication potential due to better nutrient removal; however, this is achieved at the expense of a higher levels of energy needed to treat water to lower nutrient concentrations using energy intensive technologies. For example, energy and chemical costs of a 10 mgd facility implementing nitrogen and phosphorus removal increase from $350 per million gallon (MG) for a treatment level of 8 mg N/L and 1 mg P/L to $1,370 per MG for a treatment level of 2 mg N/L and <0.02 mg P/L (WERF, 2011). However, higher levels of energy consumption make indirect sources of eutrophication potential (e.g., NO_x emissions from electricity production) more prevalent at this scale. At the community scale, 48% of the eutrophication potential comes from electricity, whereas household contributions from electricity account for 18% and city level contributions from electricity account for only 9% of the total eutrophication potential. This finding coincides with a previous study, where Foley et al. (2010) found that treating wastewater effluent to a higher quality can improve the water quality of receiving water bodies by lowering eutrophication; however, this requires higher levels of energy consumption. WWTP managers should consider this trade-off when implementing technologies for nutrient removal at different scales. In this study, the benefits of treating to nutrients to a higher level at the community scale outweigh the drawbacks of higher levels of direct nutrient emissions at the household scale, and higher nutrient emissions from land applied biosolids at the city scale.

3.4.6 Impact of Scale on Eutrophication Offset Potential of Resource Recovery

Eutrophication offsets associated with integrated resource recovery are relatively comparable at the household (0.85 g PO_4eq/m^3), community (0.81 g PO_4eq/m^3), and city (0.79 g PO_4eq/m^3) scale. This occurs because potable water offsets decrease with scale, while fertilizer offsets of eutrophication potential increase with scale, leading to an overall balance of integrated resource recovery offset potential. The significance of potable water offsets decreases with scale because the percentage of reclaimed water used decreases as centralization increases. Whereas all the water can be reclaimed at the household level through subsurface drip irrigation, 23% of the treated effluent is discharged to surface water at the community scale during the rainy season. At the city scale, approximately 44% of the effluent goes to deep well injection where there is no potable water offset or nutrient offset benefit, but water supply is replenished and salt water intrusion is prevented. Fertilizer offsets of eutrophication potential increase with level of centralization primarily because fertilizer offsets from land application of biosolids increase as biosolids production increases. Nutrient recycling leads to an increase in relative contribution of phosphorus fertilizer offsets from household (0.5%) to community (6%) to city (7%) scale. The increased significance of fertilizer offsets as scale increases, in addition to the slight offset contribution from energy recovery, leads to comparable results for eutrophication offsets at all scales. Previous studies have not considered how integrated resource recovery offsets impact eutrophication potential at different scales.

3.4.7 Uncertainty Analysis

The Monte-Carlo uncertainty analysis evaluates the uncertainty associated with embodied energy, carbon footprint, and eutrophication potential of the three systems evaluated. A normal distribution is assumed at all scales for the various material, energy, GHG emission, and nutrient emission inputs to SimaPro 8 (PhD version). The standard deviation of embodied energy at the city scale has a higher standard deviation (σ=4.8) than the community (σ=1.0) and household scale (σ= 0.4), where the greatest contributor to embodied energy at all scales is direct energy from operational electricity consumption. Operational electricity inputs for the household and community systems are based on annual averages, whereas electricity inputs for the city scale capture seasonal fluctuations. Consequently, the standard deviation is higher at the city scale because there is a wider variation in electricity inputs available at this scale, and not necessarily because data at the city scale is less certain. At the city scale, average, maximum, and minimum electricity consumption values of specific unit processes (i.e., aeration, distribution, chlorine contact chamber) from five representative months in 2013 were available. In contrast, at the household and community scale the standard deviation is lower due to a lack of data availability, not necessarily because data at these scales are more certain.

The standard deviation of carbon footprint at the household scale has a higher standard deviation (σ=0.31) than the city (σ=0.20) and community scale (σ= 0.05). At the household scale scope 1 and 3 emissions account for 45% of the carbon footprint and scope 2 emissions account for 55% of the total carbon footprint. Uncertainty is likely due to the variations in inputs associated with treatment tank infrastructure and direct CH_4 and N_2O emissions, since these are the dominant contributors to Scope 3 and 1 emissions, respectively. Treatment tank infrastructure inputs are based on the volume of concrete and mass of reinforcing steel calculated for liquid volumes of 1,000-1,050 gallons based on technical drawings and specifications from septic tank manufacturers (See Appendix A). Obtaining input data on concrete and reinforcing steel directly from contractors would likely decrease the uncertainty associated with these inputs. Direct CH_4 inputs are based EPA estimation equations that require influent and effluent BOD_5 values and assumed conversion factors (EPA, 2010). Direct N_2O emissions from wastewater require inputs on influent flow rate, influent TKN and assumed constants, whereas N_2O emitted during land application of biosolids inputs require annual amount of biosolids applied to soils and assumed constants (IPCC, 2006; Chandran, 2010; EPA, 2010). There is uncertainty related to these calculations since both seasonal and diurnal fluctuations in input parameters (i.e., BOD5, TKN, flowrate, etc.) and assumed constants vary with site-specific conditions. Consequently, direct measurements of CH_4 and N_2O may decrease the uncertainty associated with these values. At the city scale scope 3 emissions are the dominant contributor to carbon footprint largely due to an increase in chemical consumption, where chemicals are the dominant contributor to Scope 3 emissions. Input data from chlorination includes average, minimum, and maximum values of monthly chlorine usage in 2012 (n=12). The standard deviation at the city scale most likely arises from seasonal variations in chemical input data, not necessarily because data is less certain. At the community scale, the standard deviation of carbon footprint is the lowest. This is likely due to a lack of data available, in which electricity (Scope 2 emissions) is the dominant contributor at the community scale emissions and only average annual values of operational electricity were available.

Similar to carbon footprint, the standard deviation of eutrophication potential at the household scale (σ=4.34) was higher than the city scale (σ=1.48) and community scale (σ=1.13). Whereas direct sources of eutrophication are the dominant contributor to eutrophication potential at the household and city scale, indirect sources are the dominant contributor to eutrophication potential at the community scale. Consequently, the standard deviation at the household and city scale arises from variations in the range of nutrients discharged to the environment, not necessarily because data is less certain. For example, at the household scale direct nitrogen loads from reclaimed water are the dominant contributor to eutrophication potential. These average, minimum, and maximum inputs are calculated based on typical effluent concentrations from septic tanks with aerobic treatment from previous literature (Asano, 2007), accounting for plant uptake of nutrients (See Appendix A for further details). Gathering on-site data from systems directly, might decrease the uncertainty of these results. At the city scale, nitrogen from reclaimed water and biosolids are the dominant contributors to eutrophication potential. These average, minimum, and maximum values are calculated from monthly averages of nitrogen concentrations in reclaimed water and biosolids in 2012. These data provide an accurate portrayal of seasonal variation and leading to the dominant contributor to the standard deviation at the city scale. The dominant contributor to indirect sources of eutrophication at the community scale is operational electricity. Since only an average annual value was available for this scale the standard deviation was lower than the household and city systems. Consequently, increasing access to monthly electricity data, or at least electricity data that captures seasonal variations would be beneficial to increasing the certainty of results. The uncertainty analysis highlights how uncertainty can change with scale, impact category, and data availability.

3.5 Conclusions of United States Case Study

This chapter used life cycle assessment (LCA) to evaluate scale's influence on the environmental sustainability of WWTPs integrated with resource recovery at the household, community, and city levels. Tampa, FL was selected as the site location because it represents a typical urban coastal city in the developed world facing population growth, nutrient sensitive water bodies, and vulnerability to climate change impacts. Proven technologies used throughout the U.S. were selected for analysis. Embodied energy and carbon footprint were used as global sustainability indicators, whereas eutrophication potential was used to evaluate local sustainability of water to explore the impacts and trade-offs of the water-energy-carbon-nutrient nexus as it relates to wastewater management strategies.

Global impacts (e.g., embodied energy and carbon footprint) adhere to economies of scale where centralization leads to lower environmental impacts, despite fluctuations in specific trends within each impact category. Consequently, alternative household systems that implement less energy-intensive technology (e.g., household level passive nitrogen reduction methods with gravity trenches designed for optimal water reuse) may lead to more sustainable ways to treat wastewater for beneficial reuse at the decentralized level. Embodied energy and the associated carbon footprint of treatment is highest at the community scale due to higher energy usage for nutrient removal and other technologies (e.g., additional UV treatment and aerobic digestion), indicating that treatment technology in addition to scale can influence the environmental sustainability of wastewater management strategies. Whereas, higher energy usage at the community scale is beneficial to reducing local impacts (e.g., eutrophication potential), it simultaneously leads to higher global impacts (e.g. embodied energy and carbon) highlighting trade-offs between impact categories investigated. WWTPs could consider implementing energy efficient strategies (e.g., heat pumps, VFDs, energy-efficient aeration) and managing wastewater treatment differently as seasons change. For example, the community scale system could reduce global impacts by removing nutrients only during the rainy season when water is discharged to surface water bodies, but maintaining nutrients within the treated effluent to increase beneficial reuse of nutrient rich reclaimed water during the dry season. This would require regulatory changes to accommodate seasonal water reuse.

In addition, water reuse distribution has a lower impact than treatment compared to other regions (e.g., California) where topographical conditions are different. Furthermore, water reuse has the highest offset potential for both global and local impact categories, highlighting the benefits of replacing potable water with reclaimed water for irrigation purposes. In this study, Florida's flat topography appears to favor semi-centralization (community scale) or centralization (city scale) of wastewater management, particularly when energy-efficient variable frequency drive pumps are used for water reuse distribution. However, decentralization (household scale) and semi-centralization (community scale) provide higher potable water offsets than centralization (city scale), since a higher percentage of water is reclaimed for beneficial reuse at these scales.

This highlights that water and nutrient reuse may be more effective at the community scale, whereas integrated resource recovery (e.g., water reuse, nutrient recycling, and energy recovery) leads to the greatest percent offset at the city scale. Fertilizer offsets have the lowest mitigation potential for all impact categories, yet are highest at the city scale due to larger production of biosolids rich in nitrogen, highlighting benefits to centralization for nutrient recycling. Energy recovery is only applicable at the city scale, in which the integration of water reuse, energy recovery, and nutrient recycling leads to a 49% offset of embodied energy. This is approximately equivalent to all the direct energy needed for collection, treatment, and water reuse distribution. In addition, integrated resource recovery at all scales can effectively offset all of the scope 1 emissions associated with wastewater management, and at the city scale is approximately equal to all of the scope 2 emissions associated with treatment and reuse. These findings highlights that there are benefits to hybrid systems, where water is reclaimed locally, but biosolids are treated at a centralized facility. Reclaiming water locally (e.g., community scale) would increase potable water offsets, while achieving a high level of treatment for environmental and human health protection. Treating biosolids at a centralized facility (e.g., city scale) would increase fertilizer offsets from nutrient recycling and lead to beneficial energy offsets from energy recovery. The uncertainty analysis highlights how standard deviation change with scale where in some cases data availability has a larger impact on standard deviation than actual uncertainty. This highlights the importance of enabling access to data that captures seasonal variations to ensure accurate analysis of uncertainty at varying scales.

CHAPTER 4: CONTEXT'S INFLUENCE ON THE ENVIRONMENTAL SUSTAINABILITY OF WASTEWATER TREATMENT PLANTS WITH INTEGRATED RESOURCE RECOVERY

4.1 Abstract

Despite global concerns of lack of sanitation provision, water scarcity, climate change, and resource depletion, limited research has been conducted to assess the environmental sustainability of wastewater treatment and resource recovery strategies to improve access to sanitation and resource utilization in developing world settings. Furthermore, limited studies have investigated how context (e.g., rural developing world versus urban developed world) impacts the environmental sustainability of wastewater treatment plants (WWTPs) with integrated resource recovery. Accordingly, this chapter[2] seeks to evaluate the potential benefits of mitigating the environmental impact of two small community-managed wastewater treatment systems in rural Bolivia using resource recovery (i.e., water reuse, nutrient recycling and energy recovery). These systems are then compared to the United States community scale WWTP with integrated resource recovery analyzed in Chapter 3. Life cycle assessment (LCA) is used to estimate the embodied energy, carbon footprint, and eutrophication potential of these systems under existing and resource recovery conditions. Two distinct technologies are analyzed in

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[2] The majority of this chapter was reprinted from *Journal of Environmental Management*, 131/2013, Pablo K. Cornejo, Qiong Zhang, James R. Mihelcic, Quantifying benefits of resource recovery from sanitation provision in a developing world setting, 7-15, Copyright (2013), with permission from Elsevier.

Bolivia: (1) an upflow anaerobic sludge blanket reactor (UASB) followed by two maturation ponds in series (UASB-Pond system) and (2) a facultative pond followed by two maturation ponds in series (3-Pond system). To assess the impact of context, these systems are then compared to the U.S. community system consisting of primary, secondary, tertiary disinfection with UV and chlorination, and aerobic digestion.

For the existing systems in Bolivia, the results indicated that bathroom and collection infrastructure had a higher energy intensity than the treatment processes, whereas direct biogenic greenhouse gas (GHG) emissions from treatment were the primary contributors to the carbon footprint. Taking advantage of reclaimed water was found to greatly reduce the eutrophication potential for both systems, with the reduction increasing proportionally to the percentage of water reclaimed. Energy recovery from the UASB-Pond system provided a 19% reduction in embodied energy and a 57% reduction in carbon footprint. Combining water reuse with nutrient benefits and energy recovery for the UASB-Pond system reduces eutrophication potential, embodied energy and carbon footprint simultaneously. This highlights the benefits of integrated resource recovery.

In contrast, the U.S. community system was found to have a higher carbon footprint and embodied energy than the two Bolivian systems, yet a lower eutrophication potential. Whereas, high treatment levels for nitrogen removal leads to lower local impacts (e.g., eutrophication potential), higher energy usage from mechanized systems in U.S. leads to higher global impacts (e.g., embodied energy and carbon footprint), compared to systems integrating natural wastewater treatment technologies in rural Bolivia. This highlights how differences in context (e.g., location, operation and maintenance, treatment technology, resource recovery strategies, and other demographics) lead to trade-offs between the U.S. and Bolivia based systems.

4.2 Introduction

Global stressors, such as population growth, increasing urbanization, and climate change place additional pressure on already limited water resources (Zimmerman et al., 2008). For example, water demand is expected to rise as the global population increases by an estimated 32%, from 6.9 to 9.1 billion people by 2050 (Evans, 2011). Additionally, global climate change has been linked to shifting precipitation patterns and weather shocks that impact the hydrological cycle, water quality, and water supply (Bates et al., 2008).

Amidst these realities, the developing world faces unique water and sanitation challenges. A large proportion of the developing world's urbanizing population will live in small towns, where populations and the number of small towns are expected to quadruple in the next 30 years (Caplan and Harvey, 2010). Consequently, the provision of sanitation to small urbanizing towns is a key component to meeting the United Nations millennium development target 7c to "halve, by 2015, the proportion of the population without sustainable access to safe drinking water and basic sanitation" (UN, 2011).

Approximately 2-3% of the energy consumption worldwide is used to treat and transport water and in the developing world almost half of a municipal budget can be attributed to energy associated with water management (ASE, 2002). As efforts increase to treat the wastewater from around 1.5 billion people discharging through collection systems with no treatment (Baum et al., 2013), the energy consumption and greenhouse gas (GHG) emissions associated with wastewater treatment will increase as well, further contributing to climate change.

In addition to carbon and energy concerns, nutrient management of wastewater is crucial to protecting natural water bodies. More than 50% of the world's waterways are contaminated by untreated wastewater and in Latin America the majority of wastewater collected by sewer systems (85%) is not treated (Baum et al., 2013; Mara, 2004). Nutrients within the wastewater are discharged directly to nearby water bodies increasing the risk of eutrophication. Eutrophication can impair water quality by depleting oxygen levels, while harming aquatic organisms and impacting the availability of freshwater (de-Bashan and Bashan, 2004; NRC, 2012).

Nutrients, however, can be recovered from wastewater via water reuse, providing a beneficial resource to communities for non-potable uses, such as irrigation (NRC, 2012). In the developing world, irrigation demand is expected to grow with population in small urbanizing cities (<500,000 people) that rely on agriculture for local food production and economic security (Verbyla et al., 2013). Nitrogen and phosphorus recovered from wastewater can be used to increase crop yield while addressing phosphorus scarcity. In fact, an estimated 22% of the phosphorus demand worldwide can be obtained from human waste (Fatta et al., 2005; Mihelcic et al., 2011). Additionally, previous studies have found that water reuse and other types of resource recovery (i.e., energy recovery and nutrient recycling) can offset the carbon footprint of wastewater treatment systems, while reducing the utilization of fertilizers, freshwater, and fossil energy (Fine and Hadas, 2012; Mo and Zhang, 2012).

Many studies have evaluated the carbon footprint, embodied energy, and/or eutrophication potential of wastewater and resource recovery systems in the developed world (e.g., United States, Australia, Sweden, and Spain) using Life Cycle Assessment (LCA). LCA is a quantitative tool that estimates the environmental impact of a process or product over its life, including raw material extraction, construction, operation, reuse and end-of-life phases (EPA, 2006). LCA can be both labor intensive and time consuming; however, it is beneficial to reducing problem shifting by aiding researchers in identifying environmental trade-offs between impact categories, life cycle stages, and unit processes (EPA, 2006; Hendrickson et al., 2006; ISO, 2006; Miller et al., 2007; Nicholas et al., 2000). LCA has been used to investigate wastewater treatment systems, water reclamation, and energy recovery applications (Tillman et al., 1998; Hospido et al., 2004; Lundie et al., 2004; Tangsubkul et al., 2005; Ortiz et al., 2007; Meneses et al., 2010; Pasqualino et al., 2010; Mo and Zhang, 2012;) as well as water supply systems (e.g., comparing water reuse, desalination and importation or analyzing how water quality impacts embodied energy of water treatment) (Lyons et al., 2009; Stokes and Horvath, 2006, 2009; Santana et al., 2014).

In contrast, few studies have focused on the life cycle environmental impacts of wastewater systems with resource recovery outside of the industrialized world. These studies focus on larger-scale mechanized water reclamation facilities (greater than 10 mgd) serving urban areas in China (Zhang et al., 2010) and South Africa (Friedrich et al., 2009). For smaller-scale applications (<5 mgd); however, Muga and Mihelcic (2008) found that mechanized treatment technologies (e.g., activated sludge processes) are less appropriate than natural systems (e.g., waste stabilization ponds), due to higher costs and energy-intensities. Previous LCA studies have been conducted on household wastewater treatment with resource recovery in rural Peru (Galvin, 2013) and waste stabilization ponds in urban areas of Sydney, Australia (Tangsubkul et al., 2005); however, no studies have investigated the life cycle impacts of the technologies employed in this study that are appropriate for small towns in developing communities and can be integrated with resource recovery applications. Additionally, no studies identified by the author have evaluated how context (e.g., rural developing world versus urban developed world) impacts the environmental sustainability of community scale wastewater treatment plants (WWTPs) integrated with resource recovery.

Accordingly, the goal of this chapter is to evaluate the potential benefits of mitigating the environmental impact of two small community-managed wastewater treatment systems in rural Bolivia using resource recovery (i.e., water reuse and energy recovery) and compare results to community scale system in the United States, analyzed in Chapter 3. Life Cycle Assessment (LCA) is used to assess the environmental sustainability of systems under existing and resource recovery conditions using embodied energy, carbon footprint, and eutrophication potential as environmental sustainability indicators. Reclaimed water from these systems is of particular interest, because recent studies found that they have a potential to increase local food production (Verbyla et al., 2013) in a region facing population rise, increased water usage and a decrease in recharge due to climate change (Fry et al., 2012). This research provides insight to decision makers interested in improving the environmental sustainability of sanitation provision through consideration of resource recovery strategies, reclaiming water, nutrients, and energy found in wastewater.

4.3 Bolivia Case Study Background

Recent estimates indicate that 39.5% of Bolivia's population has sewer connections and only 8.3 percent of the population has sewage treatment (Baum et al., 2013). The two technologies under investigation currently treat wastewater for the rural communities of Sapecho and San Antonio in Bolivia's tropical Yungas Region. The research site location (Verbyla, 2012) is shown in Figure 12. Sapecho employs an upflow anaerobic sludge blanket reactor (UASB) followed by two maturation ponds in series (UASB-Pond system) and San Antonio employs a facultative pond followed by two maturation ponds in series (3-Pond system) (Fuchs and Mihelcic, 2011; Verbyla et al., 2013).

Both community-managed technologies have a design life of 20 years and were built in 2006. The UASB-Pond system was designed for a population of 1,471 people and has an average flow rate of 0.019 mgd (73.6 m^3/day). The 3-Pond system was designed for a population of 727 people and has an average flow rate of 0.024 mgd (91.5 m^3/day). Flow rates (n=6) were measured at both sites over a 24-hour time period during site visits from 2007 to 2012. Water committee members from both communities expressed an interest in using reclaimed water for agricultural irrigation.

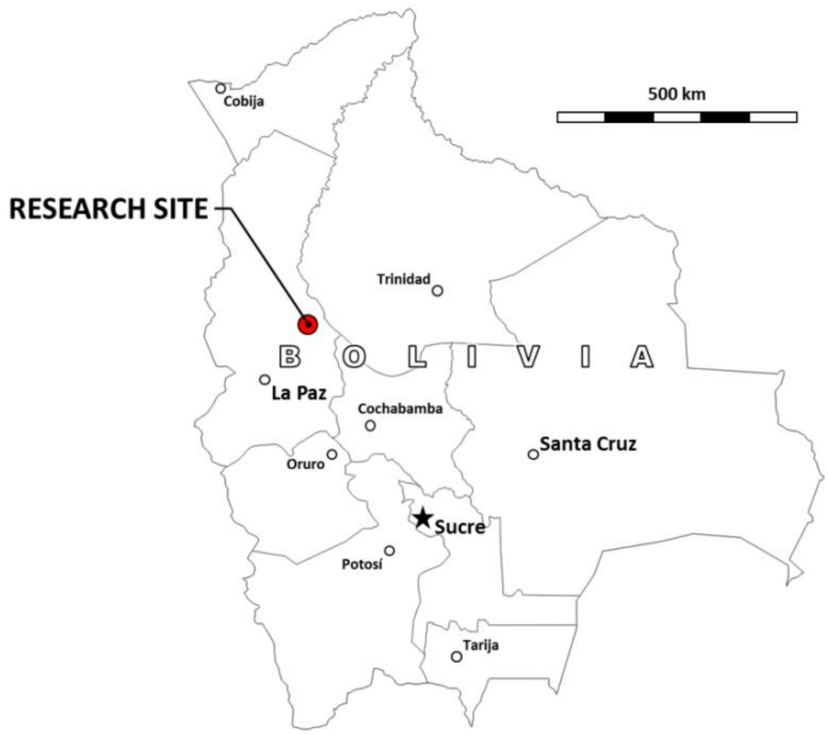

Figure 12. Bolivia research site location in Beni region. Reprinted with permission from Matthew E. Verbyla

4.4 Methods for Bolivia Case Study

To evaluate the embodied energy, carbon footprint and eutrophication potential of both systems, four steps were taken following ISO 14040 guidelines including: (1) goal and scope definition, (2) life cycle inventory, (3) life cycle impact assessment, and (4) interpretation (ISO, 2006).

4.4.1 Goal and Scope Definition

The goal of the study was to evaluate the environmental impact of the existing systems and the potential benefit of resource recovery in mitigating the impact. This is achieved by comparing embodied energy, carbon footprint, and eutrophication potential of these systems under (1) existing, (2) energy recovery, (3) agricultural water reuse, (4) and combined resource recovery (agricultural water reuse and energy recovery) conditions using LCA. Both systems were compared using a functional unit of 1 cubic meter of treated wastewater over a 20-year lifespan. Figure 13 shows the system boundaries investigated, in which construction and operation phases are considered. The existing condition includes all current unit processes for both technologies.

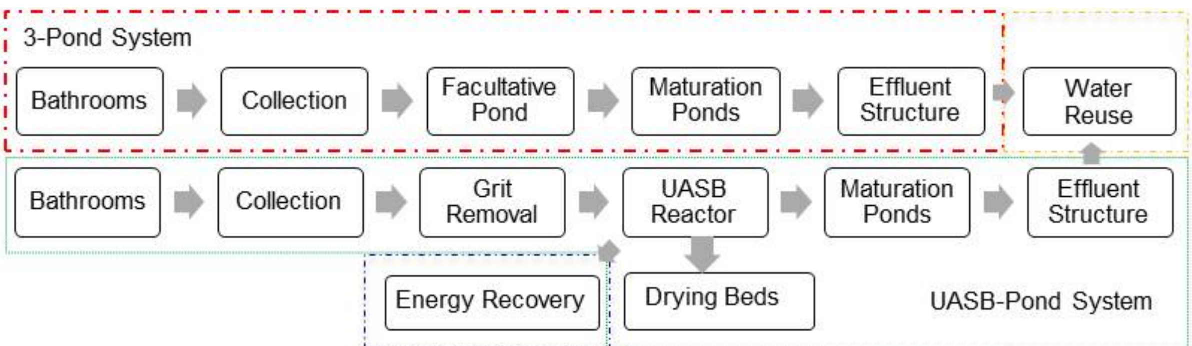

Figure 13. Boundaries for the 3-Pond system and UASB-Pond system. System boundaries include the existing condition, water reuse and energy recovery

The water reuse condition includes water reclamation for agricultural irrigation of citrus trees, through which reclaimed water provides a nutrient benefit. The benefit of the water reuse condition is quantified by comparing water reclamation to a baseline condition. Under the baseline condition, river water is used for agricultural irrigation. Under the water reuse condition, reclaimed water containing nutrients is used for agricultural irrigation. This is assumed to provide a nutrient benefit, causing an increase in crop yield by 10 to 30% (Fatta et al., 2005). The nutrient benefit is quantified by the reduction in pumping energy needed to produce an equivalent citrus yield compared to the baseline condition (river water irrigation) in this particular region. The increase in crop yield is assumed to decrease water needed to irrigate an equivalent amount of crops, therefore decreasing electricity needed for pumping irrigation compared to the baseline condition. Fertilizer offsets are not considered in the Bolivia case study because the region traditionally doesn't use synthetic fertilizers.

The energy recovery condition includes biogas recovery from the UASB reactor at the UASB-Pond site, which offsets energy consumption by avoiding the use of natural gas. The biogas was assumed to have an 65% methane composition and calculated using an EPA method (EPA, 2010). The energy offset is quantified by the amount of natural gas avoided due to the use of biogas with the same energy output (See Appendix B). Infrastructure for biogas recovery is not included in the life cycle inventory due to limited data availability. No energy recovery is possible for the 3-Pond system. Finally, the combined resource recovery condition includes both energy recovery and water reuse conditions. System expansion is used to quantify the mitigation potential associated with the resources recovered.

4.4.2 Life Cycle Inventory

Data on material production (e.g., material type, dimensions, service life and purchase frequency), material delivery (e.g., origin, weight, and transportation mode), and equipment operation/energy production (e.g., equipment type, power use, amount use, and use frequency) were obtained during a field study. A national Bolivian electricity mix of 44% fossil fuels, 54% hydropower, and 1.5% other (CIA, 2012) was used to estimate impact associated with electricity usage. For a detailed explanation of data collection, calculations, and inventory items, refer to Appendix B. The Ecoinvent database (PRéConsultants, 2008) was used for background data, such as raw materials extraction, material production and transportation, and electricity generation.

4.4.3 Life Cycle Impact Assessment and Interpretation

The impact assessment was conducted using the methods provided in SimaPro 7.2 (PRéConsultants, 2008). Three impact indicators were selected in this study: (1) carbon footprint (as global warming potential (GWP) in kilograms of carbon dioxide equivalents (kgCO_2eq)) using the Intergovernmental Panel on Climate Change (IPCC) 2007 GWP 100a method; (2) embodied energy (as cumulative energy demand (CED) in megajoules (MJ)) quantified using the Cumulative Energy Demand method (Hischier et al., 2010), and (3) eutrophication potential (EP as kilograms of phosphate equivalents (kgPO_4eq)) using Eco-indicator 95 (Goedkoop, 1995). Results for the entire system and each unit process were then interpreted to determine the embodied energy, carbon footprint, and eutrophication potential per functional unit.

These environmental impact categories were selected because of their relevance to wastewater treatment and resource recovery strategies. Previous studies found that both energy and carbon footprint are dominant contributors to the environmental impact of water reuse systems (Lyons et al., 2009; Ortiz et al., 2007). Eutrophication potential was selected because of its relevance to wastewater treatment (Hospido et al., 2004), where reclaiming water can reduce the risk of eutrophication in nearby water bodies. This study assumes both systems can be designed and operated to provide an effluent that is safe in terms of health risk (WHO, 2006). Other research has focused on pathogen removal of the two systems investigated (Symonds et al., 2014) and waste stabilization ponds worldwide (Verbyla and Mihelcic, 2015).

4.4.4 Sensitivity Analysis

A sensitivity analysis was performed to identify input parameters to which the results are sensitive, by calculating sensitivity factors (SF). Inputs with a percent contribution of 1% or lower were considered negligible. For each material, energy, or emission inventory item, the input value was modified by ±20%. Then, the embodied energy, carbon footprint and eutrophication potential of the existing system were re-calculated to determine how the change in input impacted the resulting impact category. The relative change of output was compared with the relative change of the input terms to calculate the SF.

4.5 Results and Discussion for Bolivia Case Study

4.5.1 Life Cycle Inventory Results

A comprehensive life cycle inventory of both systems can be found in Appendix B including inputs related to material, energy, transportation, and emissions. Material input parameters with process contributions greater than 1% include the amount of cement, wood, PVC, cast iron, clay brick, HDPE, sanitary ceramics, reinforcing steel, and door wood used. Energy input parameters include the amount of diesel and electricity consumed. Air emissions include biogenic CO_2 and biogenic CH_4, whereas emissions to water include total nitrogen (TN), and total phosphorus (TP).

During construction, ceramic bricks and sanitary ceramics were solely used in bathroom infrastructure, whereas cement, wood, PVC, and transportation were largely consumed during the construction of bathrooms and collection systems. Electricity and diesel consumption was highest during the construction of the collection system, but pond construction also had high diesel consumption.

During the operation phase, carbon dioxide (CO_2) and methane (CH_4) emissions from the treatment processes (e.g., biogenic emissions from the UASB reactor followed by maturation ponds, and facultative pond followed by maturation ponds) were high. Whereas CO_2 emissions are considered carbon neutral, other pertinent greenhouse gases (GHGs) for wastewater (e.g., nitrous oxide (N_2O)) have a negligible contribution for waste stabilization ponds (e.g., anaerobic ponds, aerobic ponds) (IPCC, 2006). Methane is therefore the principle GHG of concern for these systems. The UASB reactor was the largest contributor to CH_4 emissions. Other relevant operational items included transportation and diesel usage during sludge removal and geomembrane replacement for the facultative lagoon.

4.5.2 Existing Bolivian Systems

4.5.2.1 Embodied Energy of the Existing Bolivian Systems

A summary of the embodied energy as cumulative energy demand (CED) for each site is shown in Table 20. Material and energy consumption during the construction phase had a significantly higher contribution to the embodied energy than the operation phase for both systems. Dominant contributors were wood (e.g., form wood, construction wood), diesel used

Table 20. Embodied Energy and percent contribution of each unit process for Bolivia systems (3-Pond and UASB-Pond)

	3-Pond		UASB-Pond	
Unit Process	**Embodied Energy (MJ/m^3)**	**Percent Contribution (%)**	**Embodied Energy (MJ/m^3)**	**Percent Contribution (%)**
Residential Bathrooms	2.49	20.0	6.73	38.9
Collection System	6.12	49.1	7.01	40.5
Facultative Lagoon	2.62	21.0	-	-
Maturation Lagoons	1.22	9.8	2.03	11.8
UASB Reactor	-	-	1.19	6.9
Grit Removal Chamber	-	-	0.10	0.6
Sludge Drying Bed	-	-	0.11	0.6
Effluent Structure	0.01	0.1	0.12	0.7
Total	12.5	100	17.3	100

by construction equipment, and PVC. These items constituted approximately 66-77% of total embodied energy. The operation phase of the 3-Pond system only accounted for 10% of the embodied energy and was negligible for the UASB-Pond, due to the low electricity and material consumption to operate and maintain these systems integrating natural treatment processes.

The collection system and residential bathrooms make up the largest contribution of embodied energy accounting for approximately 69% of the total CED for the 3-Pond system and 79% of the total CED for the UASB-Pond system. Consequently, bathrooms and sewage collection had a more significant impact on the embodied energy than wastewater treatment processes, particularly for the UASB-Pond system. It is important to note that the embodied energy of wastewater treatment only, excluding bathrooms and collection, is low at 3.8 and 3.5 MJ/m^3 for the 3-Pond and UASB-Pond system, respectively. In contrast, the embodied energy for wastewater treatment typically used in developed world settings (e.g., activated sludge) is much higher at 13.3 MJ/m^3 (Pasqualino et al., 2010) when bathrooms and collection are excluded.

These results differ from mechanized systems typically used in developed world settings, in which large electricity consumption lead to higher embodied energy during the operation phase (Stokes and Horvath, 2006). Furthermore, residential bathrooms and collection systems are well known to be key contributors to improved health through provision of sanitation, hygiene, and the transport of pathogens away from a community. However, collection systems require energy for construction and materials to transport large quantities of water (up to 0.075 m^3/capita-day) to properly function, and can decrease downstream health and economic opportunities if the collected wastes are not appropriately managed (Fry et al., 2008). Consequently, these findings highlight that for less mechanized treatment systems in the developing world, bathrooms and collection infrastructure are not only important to improving health and addressing the global sanitation crisis, but also have important energy implications.

4.5.2.2 Carbon Footprint of Existing Bolivian Systems

Whereas the embodied energy implications were highest during the construction phase, the carbon footprint was more prevalent during the operation phase. The operation phase had a 61% and 69% carbon footprint contribution for the 3-Pond and UASB-Pond systems, respectively. A summary of the carbon footprint as global warming potential (GWP) for both sites is shown in Table 21.

Table 21. Carbon footprint and percent contribution of each unit process for Bolivia systems (3-Pond and UASB-Pond)

	3-Pond		UASB-Pond	
Unit Process	**Carbon Footprint (kg of CO_2eq/m^3)**	**Percent Contribution (%)**	**Carbon Footprint (kg of CO_2eq/m^3)**	**Percent Contribution (%)**
UASB Reactor	-	-	1.17	57.6
Facultative Lagoon	0.43	56.9	-	-
Maturation Lagoons	0.07	9.1	0.33	16.1
Bathrooms	0.10	12.6	0.28	13.7
Collection System	0.16	21.3	0.23	11.2
Pretreatment	-	-	0.01	0.3
Sludge Drying Bed	-	-	0.004	0.2
Effluent Structure	0.001	0.1	0.02	1.0
Total	0.76	100	2.0	100

The operation phase was dominant at both sites due primarily to high direct biogenic emissions from the treatment processes (e.g., UASB reactor, facultative lagoon, and maturation lagoons). The facultative and maturation lagoons in series accounted for approximately 66% of the carbon footprint for the 3-Pond system, whereas the UASB reactor and maturation lagoons in series accounted for 74% of the carbon footprint for the UASB-Pond system. Biogenic CH_4 emissions (primarily from the degradation of organic carbon in the treatment processes) had the largest contribution to the total carbon footprint. For example, in San Antonio 58% of the carbon footprint came from biogenic CH_4 emissions. Similarly, in Sapecho, 69% of the carbon footprint came from biogenic CH_4 emissions. In contrast, fossil-based GHG emissions from construction materials and fossil energy usage made up approximately 42% of the total carbon footprint for the 3-Pond system and 31% of the total carbon footprint for the UASB-Pond system.

The carbon footprint of the UASB-Pond system (2.0 $kgCO_2eq/m^3$) was higher than the 3-Pond system (0.76 $kgCO2eq/m^3$), largely due to the CH_4 emissions from the UASB reactor and maturation lagoons. These findings differ from previous studies on larger, mechanized wastewater treatment systems in both developed and developing world settings, where the operation phase is the dominant contributor to the carbon footprint, primarily due to indirect emissions from electricity consumption (Friedrich et al., 2009; Stokes and Horvath, 2006).

Furthermore, these findings are consistent with a previous study on waste stabilization ponds that found that CH_4 emissions from ponds are the dominant contributor to carbon footprint (Tangsubkul et al., 2005). In the developing world, efforts to mitigate the carbon footprint of systems integrating natural wastewater treatment processes and waste-to-energy processes serving smaller urbanizing populations similar to the systems investigated in this study, should therefore emphasize the mitigation of direct biogenic CH_4 emissions.

4.5.2.3 Eutrophication of Existing Bolivian Systems

The eutrophication potential (EP) as g PO_4eq/m^3 of the 3-Pond and UASB-Pond systems under existing conditions is shown in Figure 14. Currently, all of the treated effluent at both sites is discharged to a nearby river with no water reclamation in practice. Eutrophication potential of the 3-Pond system (34.4 g PO_4eq/m^3 wastewater treated) is slightly lower than the

UASB-Pond (51.2 g PO_4eq/m^3) due to lower levels of nitrogen and phosphorus in the treated effluent at the 3-Pond site.

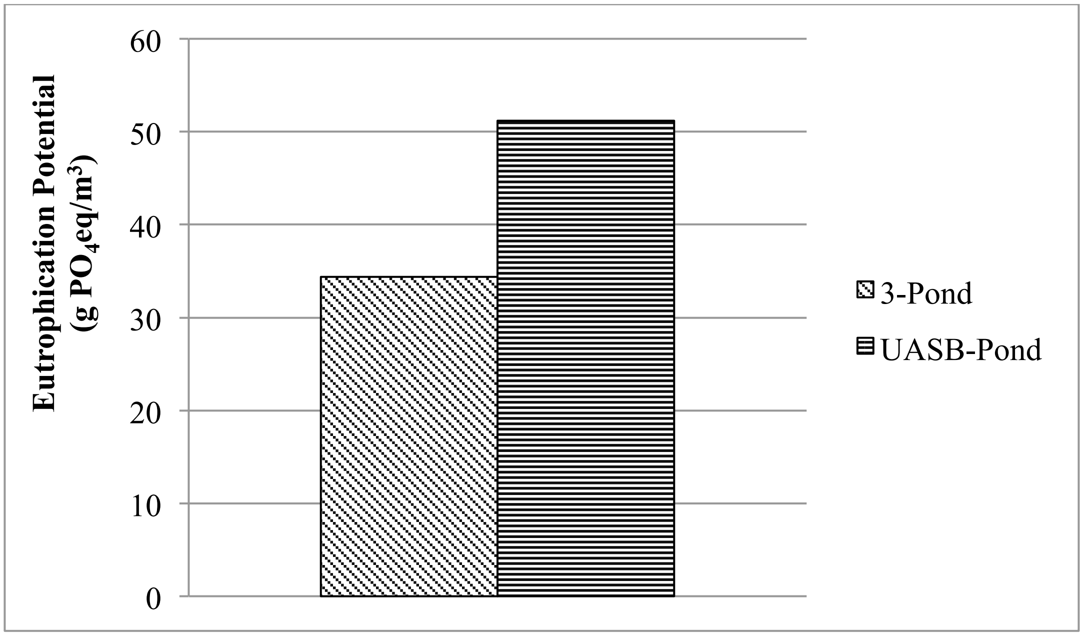

Figure 14. Eutrophication potential under existing condition for Bolivia systems (3-Pond and UASB-Pond)

The effluent concentration of total nitrogen was 51.8±28.1 mg N/L at the UASB-Pond site and 34.7±14.1 mg N/L at the 3-Pond site (Verbyla et al., 2013). Effluent concentrations of total phosphorus were approximately 9.4±4.4 mg P/L and 6.4±2.2 mg P/L at the UASB-Pond and 3-Pond sites, respectively (Verbyla et al., 2013). Total nitrogen (TN) and total phosphorus (TP) present in the treated effluent are primary contributors to the eutrophication potential, accounting for over 98% of the total impact at each site. Wood production yields the second largest contribution, accounting for only 0.2% and 0.3% of the eutrophication potential at the UASB-Pond and 3-Pond site, respectively. Cast iron and diesel production each have a 0.2% contribution at the 3-Pond site and all remaining items contributed to less than 0.1% of the eutrophication potential.

4.5.3 Water Reuse Condition in Bolivia

4.5.3.1 Embodied Energy and Carbon Footprint of Water Reuse Condition in Bolivia

The percent reduction in embodied energy as the percentage of reclaimed water utilized increases from 20% to 80% of the system's capacity and crop yield increases from 10% to 30% relative to the baseline condition is shown in Figure 15. Under the water reuse condition the embodied energy reduction potential is small, less than 2.5% for both systems, representing a maximum reduction of less than 0.3 MJ/m^3. This reduction is low because the reduction in electricity usage to pump reclaimed water is low compared to the baseline conditions (pumping of river water for crop irrigation) required to achieve the same the crop yield. As the percentage of reclaimed water and crop yield increase, the energy offset potential slightly increases. This offset potential is greatest when the maximum amount of water is reclaimed (80% of the capacity) and the maximum yield is achieved (30% increase in crop yield).

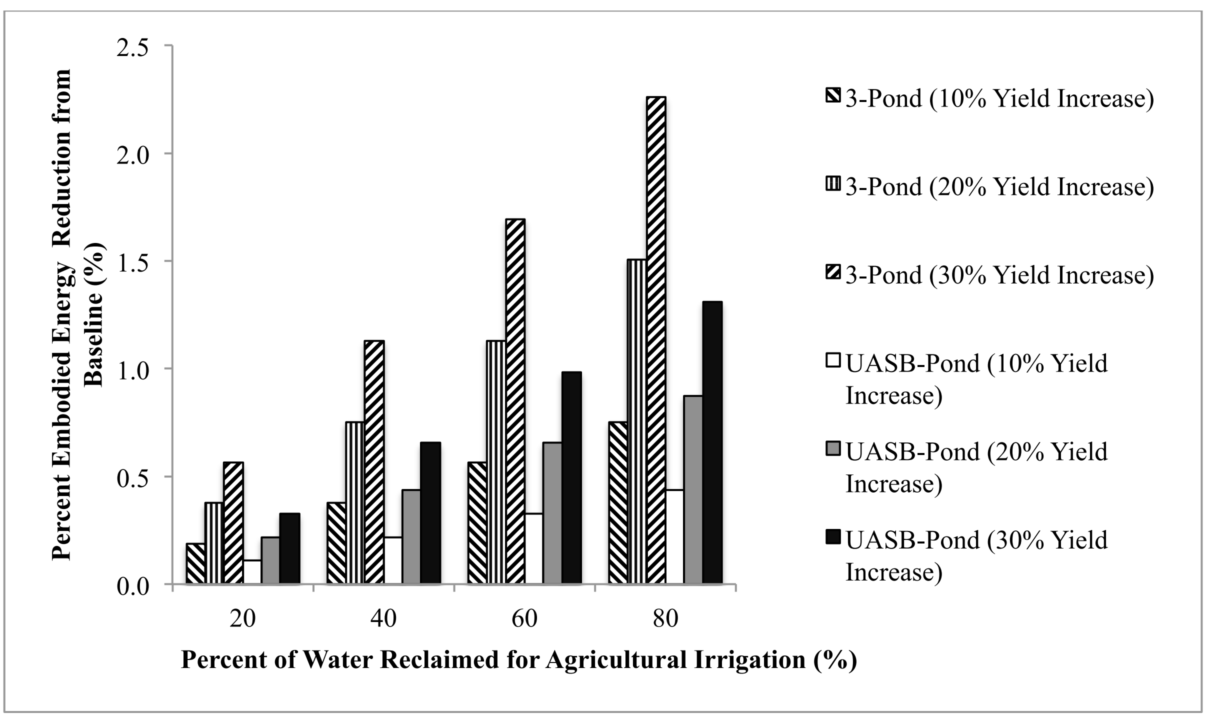

Figure 15. Percentage of embodied energy avoided as water reclamation increase from 20-80% and yield increase ranges from 10-30% for Bolivia systems

The percent reduction in carbon footprint as the percentage of reclaimed water increases (from 20 to 80% of the capacity) and crop yield increases (from 10 to 30% yield increase) is shown on Figure 16. Reclaiming water slightly decreases the carbon footprint compared to the baseline condition, less than a 3% reduction for both systems. This represents a small reduction of approximately 0.02 kg CO_2eq/m^3 under the maximum reduction condition. Similar to the embodied energy offset, the greatest reduction in carbon footprint is achieved when 80% of the treated wastewater is reclaimed and a 30% increase in crop yield is obtained. This reduction represents the highest potential offset, but is still a small contribution to mitigating the carbon footprint. The embodied energy and carbon footprint mitigation potential of water reuse from these developing world technologies is low, because the nutrient benefit provided by avoiding river water pumping to produce an equivalent amount of crops with reclaimed water is low.

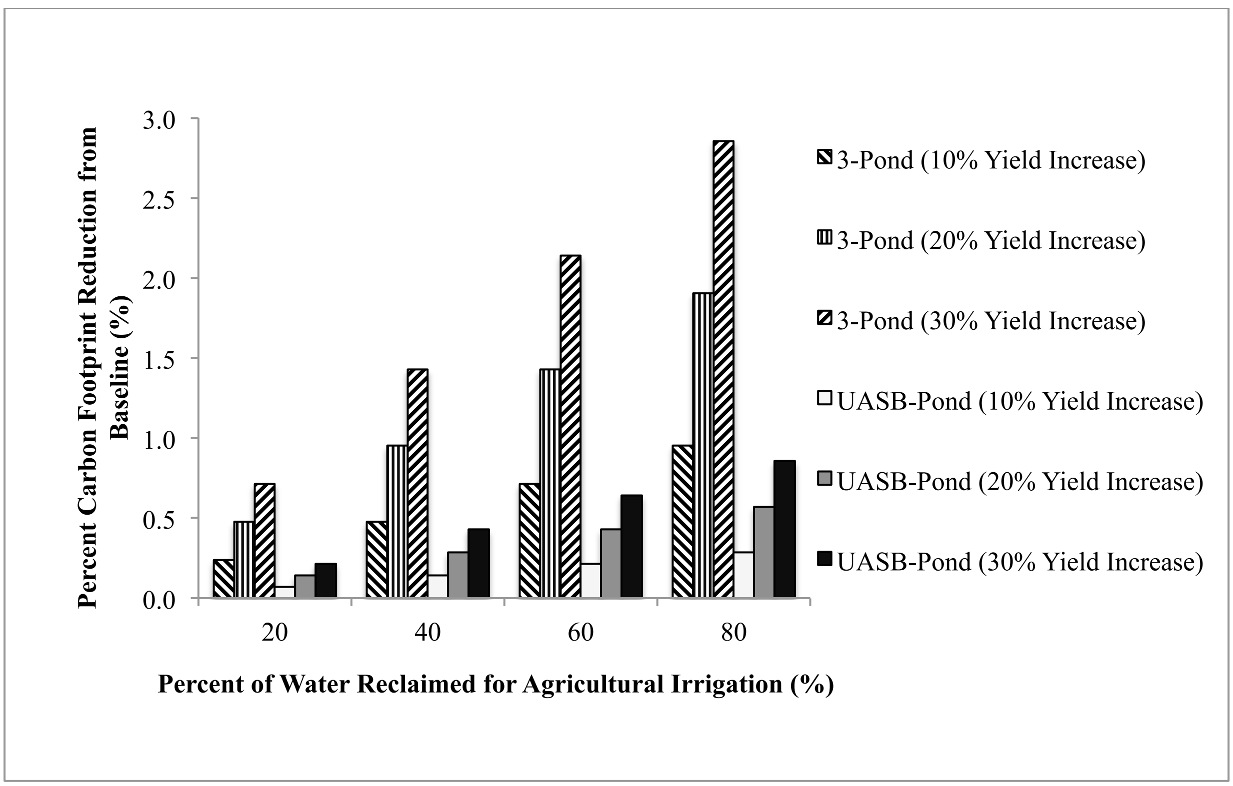

Figure 16. Percentage of carbon footprint avoided as water reclamation increase from 20-80% and yield increase ranges from 10-30% for Bolivia systems

This finding differs from a recent study on a large advanced water reclamation facility in United States (54.2 mgd average or 205,171 m^3/day), which found that water reuse has a high mitigation potential for the embodied energy and carbon footprint, with a percent offset of 37-41% and 36-40%, respectively (Mo and Zhang, 2012). This is because the benefit in the U.S. study is to avoid drinking water for irrigation and the embodied energy for drinking water is high.

4.5.3.2 Eutrophication of Water Reuse Condition in Bolivia

The percent reduction in eutrophication potential as water reclamation increases relative to baseline conditions for a 10% increase in crop yield, as shown on Figure 17. The 20% and 30% yield increase scenarios are not shown because yield increase has a minimal impact on the eutrophication potential (<0.03 g PO_4eq/m^3). This figure shows that eutrophication potential at both sites is reduced proportionally as the water reclamation increases.

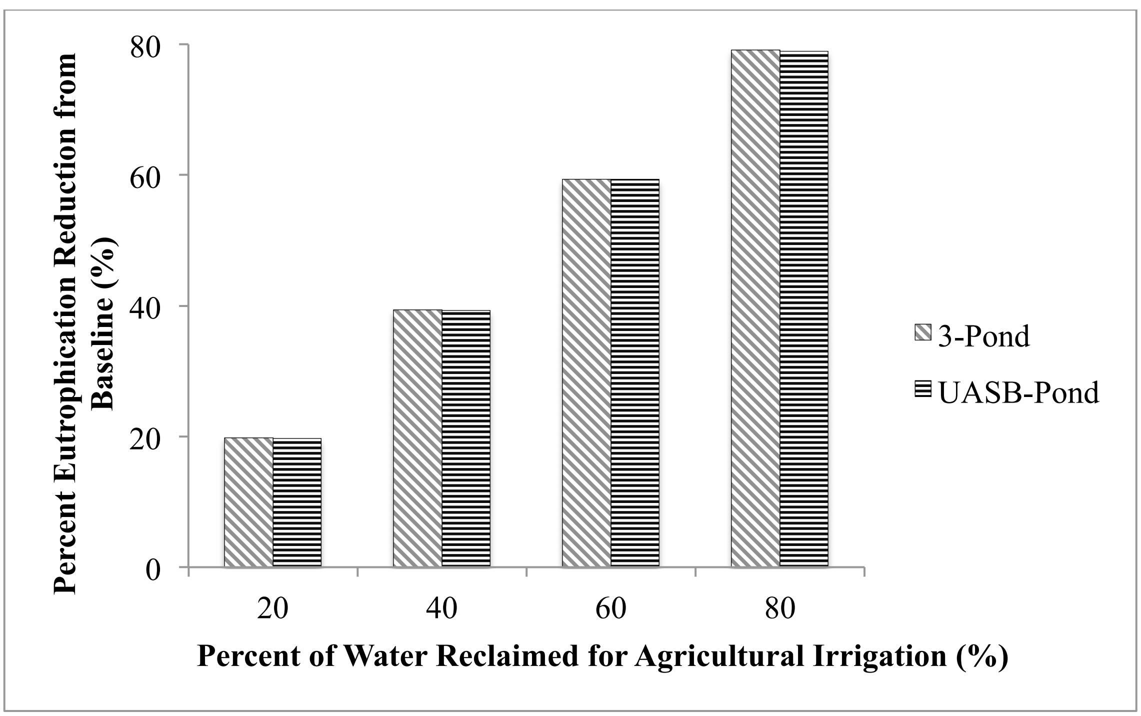

Figure 17. Percentage of eutrophication potential reduced as water reclamation increase from 20-80% for Bolivia systems

As water reclamation increase from 20% to 80% the eutrophication potential of the 3-Pond system decreases from approximately 28 to 7 g PO_4eq/m^3. Similarly the eutrophication potential of the UASB-Pond system decreases from approximately 41 to 11 g PO_4eq/m^3 as water reclamation increases from 20% to 80%. This significant reduction in eutrophication potential (19.7-79.0%) is a result of the decrease in nitrogen and phosphorus discharged to the river because in this scenario the nutrients are maintained on the land as fertilizer. Synthetic fertilizer replacement is not considered since fertilizers are not currently used in this region. The greatest mitigation potential is achieved when the maximum capacity is reclaimed (80% of the capacity), similar to embodied energy and carbon footprint under water reuse conditions, highlighting the benefits of reducing nutrient pollution when reclaiming treated water.

4.5.4 Embodied Energy and Carbon Footprint of Energy Recovery Condition in Bolivia

The embodied energy and carbon footprint for the UASB-Pond system under existing and energy recovery conditions is shown in Figure 18. The 3-Pond system has the same carbon footprint and embodied energy as the existing condition since no energy can be recovered from this site. However, energy recovery from the UASB reactor decreases the existing embodied energy from 17.2 MJ/m^3 to 14.1 MJ/m^3. This represents an 18% decrease in embodied energy, making the UASB-Pond system more comparable to the 3-Pond system under existing conditions (12.5 MJ/m^3). This reduction from existing conditions is due to the energy recovered in the form of biogas that can offsets embodied energy by avoiding the use of natural gas.

In terms of carbon footprint, the UASB-Pond system with energy recovery achieves a high reduction potential compared to the existing condition. This is a result of the avoided GHG emissions emitted from the UASB reactor when biogas is recovered. The carbon footprint for the UASB-Pond under the energy recovery condition is approximately 57% less than the UASB-

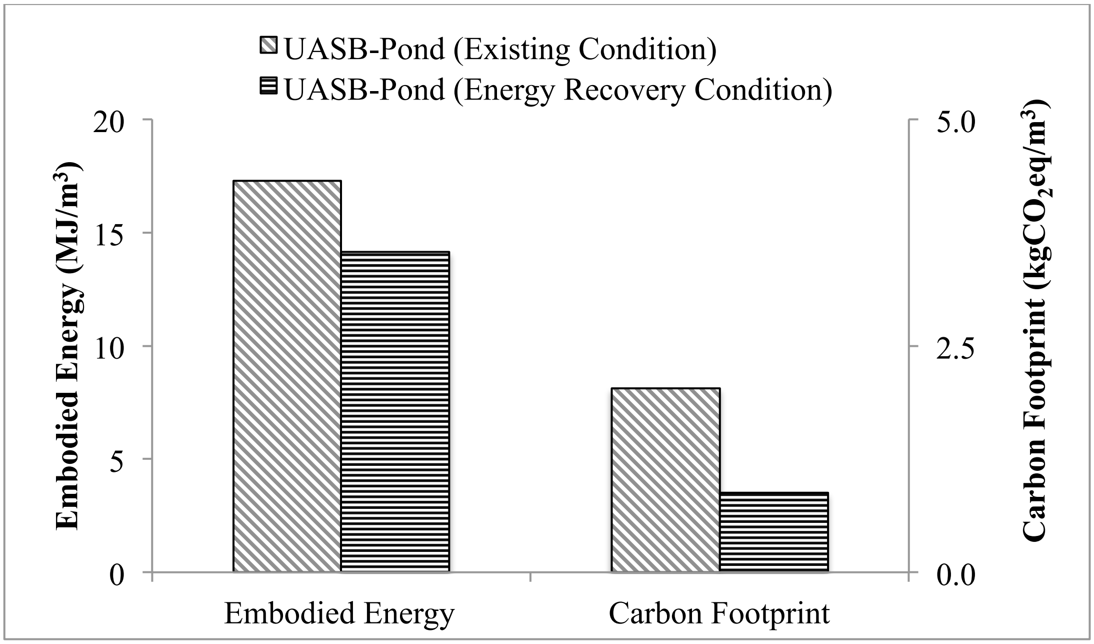

Figure 18. Embodied energy and carbon footprint under existing and energy recovery conditions for the UASB-Pond System in Bolivia

Pond under the existing condition. This makes the carbon footprint of the UASB-Pond system ($0.88\ kgCO_2eq/m^3$) under energy recovery conditions comparable to the 3-Pond system under existing conditions ($0.76\ kgCO_2eq/m^3$). This highlights the benefits of waste-to-energy processes, such as the UASB reactor, that utilize anaerobic treatment to recover biogas while mitigating the embodied energy and carbon footprint associated with natural gas production.

Certain challenges; however, may limit the recovery of biogas in actual practice (e.g., life cycle cost of infrastructure, the lack of operational capacity leading to failed systems, the low production rate and quality of the biogas, and the remote location of the UASB reactor away from the town) particularly in rural developing regions (Bruum et al., 2014). Combined heat and power (CHP) is not cost-effective at this scale (EPA, 2007; Mo and Zhang, 2013); however, a potential application in this setting is to recover the biogas as a heating fuel (Galvin, 2013). Another, perhaps more suitable option for this particular site location is flaring. The carbon footprint can be reduced through flaring, which may be a more feasible alternative than energy recovery, due to lower operation and maintenance requirements. Flaring can offset 54% of the carbon footprint compared to the existing condition; however, this provides no energy benefit. The carbon footprint offset is primarily due to the reduced UASB biogenic CH_4 emissions. Eutrophication potential under the energy recovery condition remains the same since no water is reclaimed and nutrients are still discharged to the river.

4.5.5 Summary of Combined Resource Recovery Condition in Bolivia

The percent reduction of resource recovery strategies relative to baseline conditions for water reuse, energy recovery and combined resource recovery conditions (water reuse with nutrient benefits and energy recovery) in Bolivia is shown on Table 22. Only the UASB-Pond system benefits from combined resource recovery, since energy recovery is not possible for the 3-Pond system.

Table 22. Percent reduction of resource recovery strategies from baseline condition

Condition	Embodied Energy (%)		Carbon Footprint (%)		Eutrophication Potential (%)	
	3-Pond	UASB-Pond	3-Pond	UASB-Pond	3-Pond	UASB-Pond
Water Reuse	0.2-2.3	0.1-1.3	0.2-2.9	0.1-0.9	19.8-79.2	19.7-79.0
Energy Recovery	N/A	18.2	N/A	56.7	N/A	0.03
Combined Resource Recovery	N/A	18.3-19.6	N/A	56.7-57.5	N/A	19.7-79.0

This table highlights that combining water reuse, nutrient recycling (incorporated in water reuse offset) and energy recovery at the UASB-Pond site provides a reduction in embodied energy, carbon footprint, and eutrophication potential. Energy recovery has the largest mitigation potential on the embodied energy and carbon footprint. Combining water reuse and energy recovery leads to an 18.3-19.6% reduction in embodied energy and a 56.7-57.5% reduction in carbon footprint, primarily due to energy recovery.

Water reuse is the primary contributor to offsetting the eutrophication potential. This leads to an offset of eutrophication potential of approximately 19.7-79.0% as water reclamation increases from 20-80%, where energy recovery has little effect on this impact category. Combining water reuse and energy recovery can lead to improvements in energy, carbon, and nutrient management at the UASB-Pond site. Therefore, integrating waste-to-energy technologies and water reclamation can lead to improvements in all three environmental impact categories at the UASB-Pond site.

4.5.6 Sensitivity Analysis of Bolivia Case Study

The majority of the inventory inputs have a minimal impact on the embodied energy (CED), carbon footprint (GWP), and eutrophication potential (EP) indicated by a small SF value (Table 23). Few inventory items had a large SF, indicating that these results were more sensitive to changes in input values. For embodied energy, sensitive items included the amount of wood, PVC, and diesel. This may be due to their high contribution to the embodied energy (approximately 66-77%). Diesel usage is estimated based on the equipment use hours and an hourly fuel consumption rate. Future studies can refine these input values by obtaining detailed data on actual diesel usage of specific equipment.

The carbon footprint results are most sensitive to biogenic methane emissions. This is because CH_4 emissions from the UASB reactor and facultative lagoon are the dominant contributors to the carbon footprint. Methane emissions are calculated based on BOD_5 of wastewater influent, flowrate data collected, and assumed constants (e.g., biogas composition) given by an EPA estimation method (EPA, 2010). Therefore, results can be improved by increasing data collection to assure the accuracy of these parameters.

Table 23. Sensitivity analysis results for embodied energy and carbon footprint at both sites for major inventory items based on ±20% change in input value

Input Parameters	3-Pond			UASB-Pond		
	S.F. of CED	S.F. of GWP	S.F. of EP	S.F. of CED	S.F. of GWP	S.F. of EP
Cement	0.02	0.10	0.00	0.06	0.11	0.00
Sawn Timber	0.35[a]	0.08	0.00	0.24[a]	0.03	0.00
PVC	0.10	0.06	0.00	0.22[a]	0.07	0.00
Diesel	0.33[a]	0.07	0.00	0.20[a]	0.02	0.00
Transport	0.03	0.02	0.00	0.03	0.01	0.00
Cast Iron	0.04	0.05	0.00	0.02	0.01	0.00
Clay Brick	0.04	0.02	0.00	0.08	0.02	0.00
HDPE	0.03	0.01	0.00	0.01	0.00	0.00
Ceramics	0.02	0.02	0.00	0.04	0.02	0.00
Electricity	0.01	0.01	0.00	0.02	0.02	0.00
Biogenic CO_2	0.00	0.00	0.00	0.00	0.00	0.00
Biogenic CH_4	0.00	0.46[a]	0.00	0.00	0.67[a]	0.00
TN	0.00	0.00	0.46[a]	0.00	0.00	0.47[a]
TP	0.00	0.00	0.52[a]	0.00	0.00	0.52[a]

[a]High sensitivity values. S.F. = sensitivity factor; CED = cumulative energy demand; GWP = global warming potential; EP = eutrophication potential; TN = total nitrogen; TP = total phosphorus

Total nitrogen (TN) and total phosphorus (TP) were the largest contributors to eutrophication potential (EP), accounting for more than 98% of the total. This highlights why TN and TP are sensitive to changes in input values. Continuous monitoring of TN and TP would contribute to the increased accuracy of the eutrophication potential estimations.

4.6 Conclusions of Bolivia Case Study

This study assessed the environmental impact of two community-managed wastewater treatment systems in rural Bolivia to investigate the most appropriate management strategies to integrate sanitation provision and resource recovery. Embodied energy, carbon footprint, and eutrophication potential were considered, assuming both systems treat wastewater to suitable water reuse standards for human health protection.

The embodied energy of the construction phase was found to be significantly greater than the operation phase. This resulted from a high embodied energy associated with bathroom and collection system infrastructure, compared to the treatment processes. These results revealed that the relative contribution of less mechanized wastewater treatment systems in the developing world is quite different from highly mechanized wastewater treatment technologies in the developed world. In the developing world, the inclusion of bathroom and collection infrastructure has important energy implications for the provision of environmentally sustainable sanitation.

Alternatively, the carbon footprint of the operation phase was found to be greater than the construction phase. Dominant contributors to the carbon footprint were direct biogenic CH_4 emissions from the treatment processes. This also differs from mechanized systems in the developed world, in which the production and consumption of electricity during the operation phase typically dominates the carbon footprint.

Under water reuse conditions, the nutrients diverted to land through agricultural irrigation were found to significantly reduce the eutrophication potential for both systems. This reduction increases proportionally as the amount of reused water increases, highlighting the benefit of reclaiming nutrients in treated water at both sites to reduce nutrient pollution.

However, water reuse for these systems had a low mitigation potential for embodied energy and carbon footprint compared to the baseline condition (pumping river water for irrigation). This was due to the low impact associated with reducing the electricity usage to pump reclaimed water containing nutrients, compared to pumping river water to achieve an equivalent crop yield. This finding differs from a previous study on advanced water reclamation systems in the developed world in which the benefit is the avoidance of drinking water for irrigation.

Energy recovery from the UASB reactor provided a high reduction in embodied energy and carbon footprint. This was primarily due to the natural gas avoided from biogas utilization and the offset of biogenic CH_4 emissions. By recovering energy from the UASB reactor, the UASB-Pond can achieve a comparable carbon footprint to the 3-Pond system. This points to the need to plan for usage of biogas produced in a UASB reactor (or at a minimum constant flaring of the biogas during operation).

Under existing, water reuse, and energy recovery conditions the 3-Pond system in this study was found to have a lower embodied energy, carbon footprint, and eutrophication potential than the UASB-Pond system in this particular setting. However, combined resource recovery (water reuse and energy recovery) for the UASB-Pond system was found to provide benefits in reducing the embodied energy, carbon footprint, and eutrophication potential. This highlights the benefits of integrating waste-to-energy processes with water reclamation. The current study focused on energy, carbon, and nutrient aspect of resource recovery strategies, whereas other factors such as pathogen removal, effluent quality, cost, access, and operation and maintenance should also be considered to ensure sustainability of technologies appropriate to small towns and cities throughout the developing world.

4.7 Comparison between Bolivia and U.S. Systems Investigated

Comparing community scale technologies in Bolivia and the United States requires an understanding of differences in context (Refer to Chapter 3 for detailed analysis of the U.S. community system). Context consists of a wide range of factors including: socio-politics conditions, regulations, decision-making processes, economics, and social acceptance in a given region. In this research, context refers to location-specific factors that impact wastewater management strategies including location, operational requirements, treatment technologies selected, resource recovery strategies implemented, and other pertinent demographic information as seen in Table 24. The technologies selected are largely based on location (e.g., rural versus

Table 24. Comparison of context, operation, technology, resource recovery and other demographics for Bolivia and U.S community-scale systems

Category	Name	3-Pond	UASB-Pond	U.S. Community
National Data	Location	Developing World	Developing World	Developed World
	Country	Bolivia	Bolivia	United States
	Access to Sewers[a]	39.50%	39.50%	100%
	Access to WWTP[a]	8.30%	8.30%	100%
Operation and Maintenance	Management	Community	Community	Private
	Funding	NGO and community	NGO and community	Private and Community
	Operator Skill	Low	Moderate	High
	No. of Operators	~1-2	~1-2	~4-6
	Sludge Removal	Every 2-15 years[b]	2-4 weeks	2-4 weeks
Treatment technology	Scale	Community	Community	Community
	Setting	Rural, small town	Rural, small town	Urban, gated-community
	Technology	Less mechanized, proven	Less mechanized, proven	Mechanized, proven
	Description	Facultative pond, two maturation ponds	UASB reactor, two maturation ponds	Primary, secondary, nitrification/denitrification, disinfection (UV and chlorination), aerobic digestion
Resource Recovery	Water Reuse	Agricultural irrigation to replace surface water irrigation	Agricultural irrigation to replace surface water irrigation	Golf course irrigation to replace potable water irrigation
	Energy Recovery	N/A	Biogas recovery from UASB	N/A
	Nutrient Recycling	Nutrient benefit from water reuse reduces water usage	Nutrient benefit from water reuse reduces water usage	Nutrient benefit from water reuse and biosolids replaces fertilizers
Other Demographics	Population equivalent (p.e.)	1,471	727	1,500
	Wastewater generated (gal/person/day)	16	26	207
	Population density (p.e./mi^2)[c]	5.1	5.1	722

[a]National data from Baum et al. (2013); [b] Oakley et al. (2012); [c]Bolivia data based on the population in the Beni region (INE, 2012). United States data based on population density in New Tampa (Florida Center for Community Design and Research, 2012)

urban), operation and maintenance requirements (e.g., operation skill level needed), available funding from governmental or non-governmental agencies, and wastewater management structure. These factors lead to differences in technologies appropriate for both regions.

The wastewater treatment systems in Bolivia serve small towns in rural areas located directly near agricultural areas. Given the rural context of these Bolivian communities, natural systems that require minimal training for operation and maintenance and minimal energy inputs are a preferred choice of technology (Fuchs and Mihelcic, 2011; Verbyla et al., 2013). Natural systems, such as waste stabilization ponds are more appropriate for rural developing regions land area available and limited funding for energy-intensive operation and maintenance. Natural systems primarily rely on natural physical, biological, and chemical processes to reduce organic loads and pathogen levels through natural sunlight for UV disinfection, wind for mixing and natural aeration, and solids settling in ponds with large retention times. The technologies selected include a UASB reactor (waste-to-energy system) followed by natural systems that consist of two maturation ponds (UASB-Pond) and a natural system that consists of a facultative pond followed by two maturation ponds (3-Pond). The construction cost of the 3-Pond system was $148,179, whereas the construction cost of the UASB-Pond system was $286,275, where further details on capital cost, cost/capita, training funds, water requirements, access, and management are available in previous literature (Fuchs and Mihelcic, 2011). Local water committees manage these systems and charge community members a small monthly fee for wastewater treatment services. Consequently, these systems are managed and funded by the community with some assistance from local non-governmental organizations (NGOs) (Cairns, 2014). Some technical assistance is provided by the local non-governmental organization that designed these systems; however, water quality regulations are not strictly enforced by local or national governmental agencies. This study assumes that Bolivia systems are in compliance with World Health Organization guidelines for safe reuse (WHO, 2006), where previous studies have investigated water quality issues of these systems (Verbyla et al., 2013).

In contrast, the U.S. community system serves a gated community in an urban area near a golf course. Given the population density of an urban developed world context, less land is available for treatment, requiring mechanized treatment systems with lower retention times and lower land footprints for treatment. In this context, the U.S. community system relies on energy-intensive, mechanized wastewater treatment (primary, secondary, nitrogen removal, disinfection via UV and chlorination, filtration and aerobic digestion) commonly used in urban settings in the developed world. Electricity requirements come from aeration during secondary treatment, nitrogen removal, aerobic digestion, UV, and pumping. Consequently, the U.S. community system requires a team of highly trained workers to operate and maintain the system, whereas the two Bolivia systems require less skilled workers for operation and maintenance of the waste stabilization lagoon based systems. The U.S. community system is funded through monthly fees charged to the community and is privately owned and operated by a wastewater management company. Cost information wasn't available for this system; however, the infrastructure and resource investments are typically higher for more advanced mechanized treatment systems, compared to systems integrating natural treatment processes. Additionally, the U.S. system has a higher operational cost than the Bolivia systems due to higher energy usage and more strictly enforced regulations. The U.S. systems are must meet nutrient criteria for surface water discharge at the State and national level. More stringent reinforcement of water quality standards in U.S leads to the implementation of more advanced treatment for nutrient removal, as well as other conventional parameters (i.e., pathogens, BOD_5, TSS).

Another key factor that varies between community scale systems in the U.S. and Bolivia is resource recovery strategies. For example, the UASB-Pond system is the only system with the potential for energy recovery at this scale. Biogas recovery would require additional operation and maintenance to use the biogas as a heating fuel. Additional operational capacity consisting of trained personnel would be needed to recovery biogas from this systems, various factors may lead to difficulties in implementing a sustainable biogas recovery plan (i.e., cost, operator skill level, system size, etc.). Social acceptability issues and regulatory frameworks could also be a challenge to the implementation of biogas recovery, since these issues are typically context-specific. Consequently, flaring is the current practice at the UASB site. This practice has a low implementation cost and requires a low skill operator; however, it does require consistent daily maintenance. Agricultural reuse is considered for both Bolivian technologies (UASB-Pond and 3-Pond system) due to the close proximity to agricultural areas and the community's interest in water reclamation. Agricultural reuse replaces river water irrigation, where nutrient benefits associated with reclaimed water are considered. Agricultural reuse increases crop yield and reduces energy required for irrigation compared to the current practice of river water irrigation. Fertilizer offsets are not considered in Bolivia, since these communities grow agricultural products organically, without synthetic fertilizers. Additionally, nutrient recycling from biosolids land application is not considered in Bolivia, due to the low frequency of sludge removal at the 3-Pond site and the potential health hazards associated with reclaiming untreated sludge at both sites (Verbyla et al., 2013).

In contrast, the U.S. system has no energy recovery available at this scale of implementation. Water reclaimed from the U.S. community system is used for golf course irrigation in the gated community, replacing potable water produced from the City of Tampa.

Additionally, nutrient recycling in the U.S. context comes from both reuse of water and biosolids. In the U.S. context, biosolids are treated to a level that is safe for land application. Therefore, they likely pose less of a risk to human health compared to land application of untreated biosolids from the systems in Bolivia.

Demographic information from the communities served by the wastewater treatment systems also varies between developed and developing world settings. The population served by the UASB-Pond (1,471 people) and U.S. community system (1,500 people) is comparable, whereas the 3-Pond serves less people (727 people). Additionally, the population served in Bolivia generates substantially less wastewater when normalized per person per day (an estimated 16 gal/person/day treated at the 3-Pond site and 26 gal/person/day treated at the UASB-Pond site) compared to the U.S. community system (an estimated 207 gal/person/day of wastewater generated). This difference may be due to variations in water usage in developing and developed world settings, where water usage in U.S. is substantially higher. Another factor impacting wastewater generation is population density, where there are vast differences between rural developing communities and urban developed communities. Population density impacts proximity to population served, where higher population densities often require treatment closer customers. This could possibly lead to reductions in the distance for collection of wastewater and distribution of reclaimed water. In Bolivia's rural Beni region, the population density is 5.1 people/mi^2 (INE, 2012). In contrast, the U.S. community system in New Tampa serves an urban population with a population density of 722 people/mi^2 (Florida Center for Community Design and Research, 2012). These differences in location, operational requirements, treatment technology, resource recovery and other demographics are important to consider, when analyzing the influence of context on embodied energy, carbon footprint, and eutrophication potential for WWTPs with integrated resource recovery.

4.7.1 Impact of Context on Embodied Energy

The total embodied energy of community-scale systems investigated in Bolivia and United States are shown in Figure 19. This table highlights that the total embodied energy of the UASB-Pond system and the 3-Pond system in Bolivia is lower than the total embodied energy of the U.S. community system by a factor of 2-2.7. The Bolivia systems have a lower embodied energy, primarily because they integrate natural wastewater treatment technology with minimal requirements for electricity applicable to rural developing world setting. In contrast, the U.S. community system has a higher total embodied energy because this technology is a more energy-intensive, mechanized wastewater treatment technology applicable to an urban developed world setting.

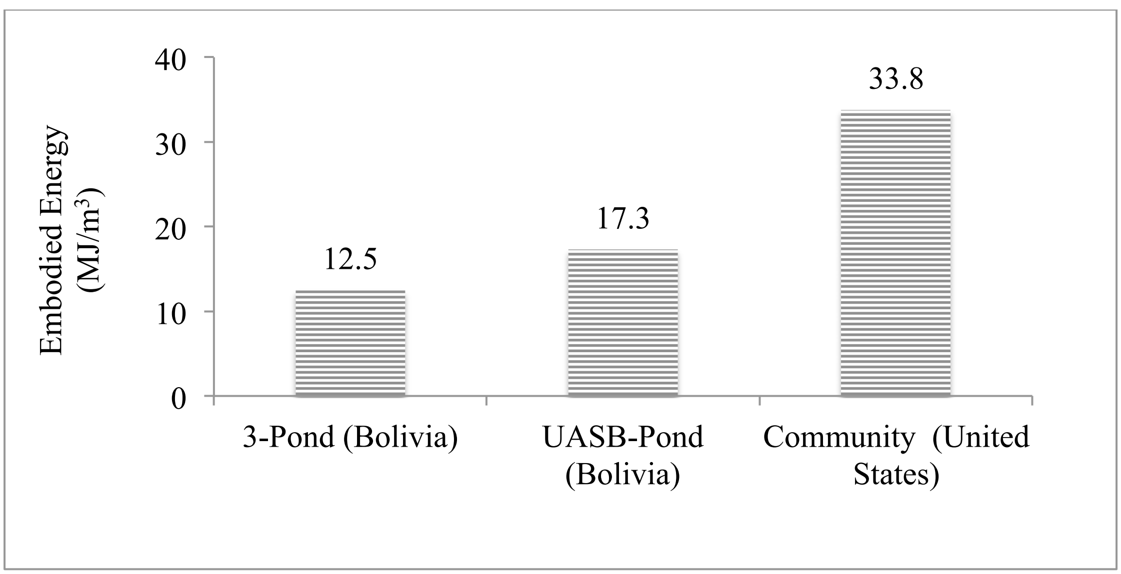

Figure 19. Embodied Energy of community-scale wastewater treatment systems in rural Bolivia (3-Pond and UASB-Pond system) and urban United States context (U.S. community system)

A key difference between both settings lies in the relative contributions from the embodied energy of collection and treatment (See Figure 20). The embodied energy of wastewater collection has a higher contribution than treatment for the community-scale systems in Bolivia, whereas the embodied energy of treatment has a higher contribution than collection for the U.S. community system. This is because of differences between technologies appropriate for rural areas in Bolivia and urban areas in United States in addition to other factors, such as population density.

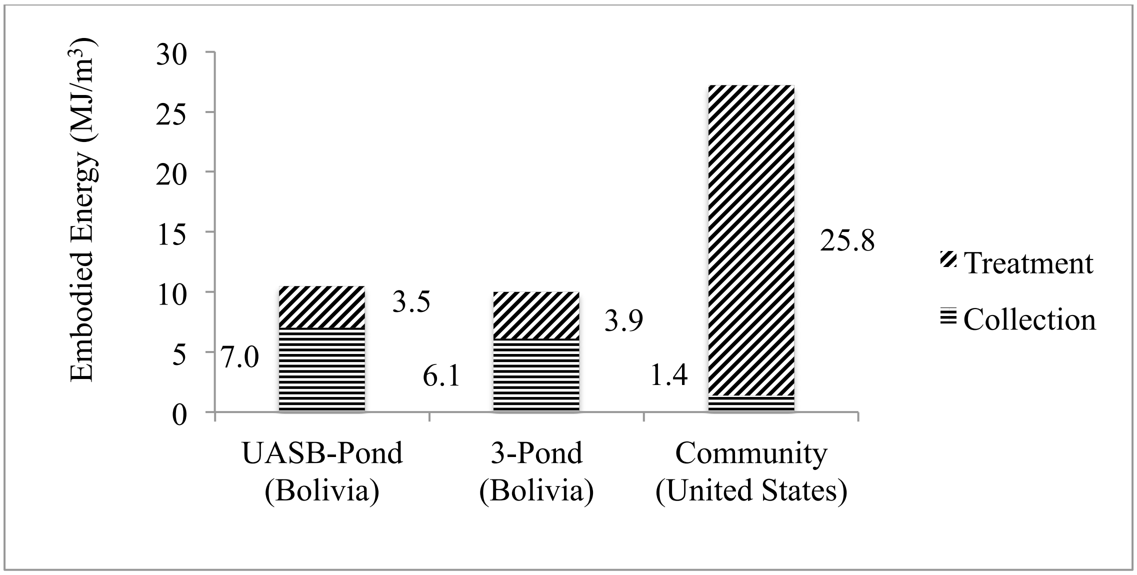

Figure 20. Embodied energy of treatment and collection for wastewater treatment systems in Bolivia (UASB-Pond and 3-Pond system) and United States (U.S. community system)

Rural areas in the developing world tend to have lower population densities (e.g., 5.1 persons/mi^2 in Beni region of Bolivia), which can possibly lead to higher collection distances for an equivalent population served or equivalent volume of wastewater treated. In addition, technologies implemented in rural areas require more land space (e.g., waste stabilization ponds) and are often implemented at further distances away from the community to ensure human health and safety. In contrast, urban areas in the developed world serve densely populated areas (e.g., 722 persons/mi^2 in New Tampa), in which wastewater treatment often occurs closer to the population served since less land area is available. In U.S., higher levels of treatment lead to lower retention times and smaller land footprints needed for treatment. Additionally, because treatment in U.S. urban areas often occurs closer to the population served, the contribution from the embodied energy of collection is lower. As a result, collection is a larger contributor to embodied energy in Bolivia's low population density regions (6.1-7.0 MJ/m^3) and a smaller contributor to embodied energy in high population density regions of the United States (1.4 MJ/m^3).

This differs from the embodied energy of treatment, where systems in a U.S. urban context have a larger contribution from treatment than systems in rural Bolivia. In United States, higher levels of treatment are implemented to meet more stringent regulations. Energy-intensive mechanized treatment technologies lead to a higher embodied energy of treatment in U.S. (25.8 MJ/m^3), compared to less mechanized systems that integrate natural treatment technologies in Bolivia (3.5-3.9 MJ/m^3). The U.S. community system also requires higher treatment levels for nutrient removal leading to higher energy usage. In contrast, Bolivia's treatment technologies are not designed for nutrient removal and therefore do not utilize energy-intensive aeration needed for nitrification. With this said, managing nutrient levels in wastewater effluent can be valuable if treatment levels match end use applications (e.g., reclaiming nutrient-rich effluent for agricultural irrigation), particularly in rural developing regions where energy-intensive treatment technologies are less appropriate.

4.7.2 Impact of Context on Carbon Footprint

Similar to embodied energy, context also has an impact on the carbon footprint of community scale wastewater treatment technologies in U.S. and Bolivia as shown in Figure 21. The U.S. community system (2.1 kg CO_2eq/m^3) has a larger carbon footprint than the systems in Bolivia. This same trend between U.S. and Bolivia systems holds true when calculating the carbon footprint per population equivalents. Despite changes in the resulting magnitude,

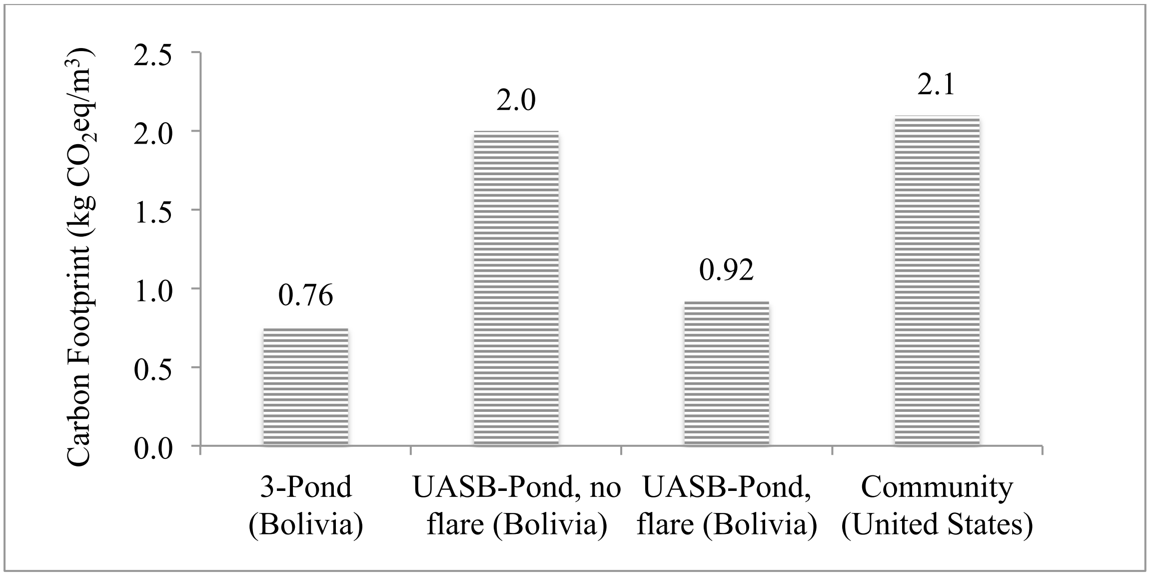

Figure 21. Carbon footprint of community scale systems in Bolivia (3-Pond, UASB-Pond with and without flare) and the United States (U.S. community system)

differences in wastewater generated and changes population served, the Bolivia systems have a lower carbon footprint than the U.S. community system when expressing results in kilograms of CO_2eq per population equivalent. The 3-Pond system has the lowest carbon footprint (0.76 kg CO_2eq/m^3) and the UASB-Pond system without flaring methane emissions from the UASB-Pond system has a carbon footprint (2.0 kg CO_2eq/m^3) comparable to the U.S. community system. This highlights the importance of the operational practice of flaring, which leads to a decrease in carbon footprint from the UASB-Pond system (0.92 kg CO_2eq/m^3), by converting CH_4 to biogenic CO_2, which is considered to be carbon neutral (IPCC, 2006).

For the Bolivia systems, direct (Scope 1) emissions are a large contributor to carbon footprint, whereas indirect (Scope 2) emissions have higher contribution for the U.S. community system (See Figure 22). Direct emissions from the 3-Pond system and UASB-Pond system without flaring contribute to 58% and 69% of the total carbon footprint, respectively. This is primarily due to CH_4 emitted from the ponds and the UASB reactor, where indirect contributions from electricity (Scope 2 emissions) are low. Flaring at the UASB site reduces the relative contribution of direct emissions from 69% to 32% of the total carbon footprint, highlighting the benefits of flaring to mitigate the carbon footprint of the UASB-Pond system. Given the high contribution from direct emissions from the Bolivia systems, mitigation efforts should focus on using natural systems without anaerobic treatment processes (e.g., 3-Pond system), anaerobic treatment systems that implement consistent flaring, or anaerobic treatment systems that take advantage of energy recovery (e.g., Galvin, 2013). This differs from mitigation efforts for the community scale U.S community system, where the contributions from direct emissions are low (only 5%). Since indirect (Scope 2) emissions are dominant contributors to carbon footprint for the U.S community system, mitigation efforts should focus on reducing electricity consumption. This can be done through the implementation of more efficient pumps with variable frequency drive (VFD), energy-efficient aeration, and waste heat recovery using a heat pump (Neuberger and Weston, 2012; EPA, 2013b; Mo and Zhang, 2013).

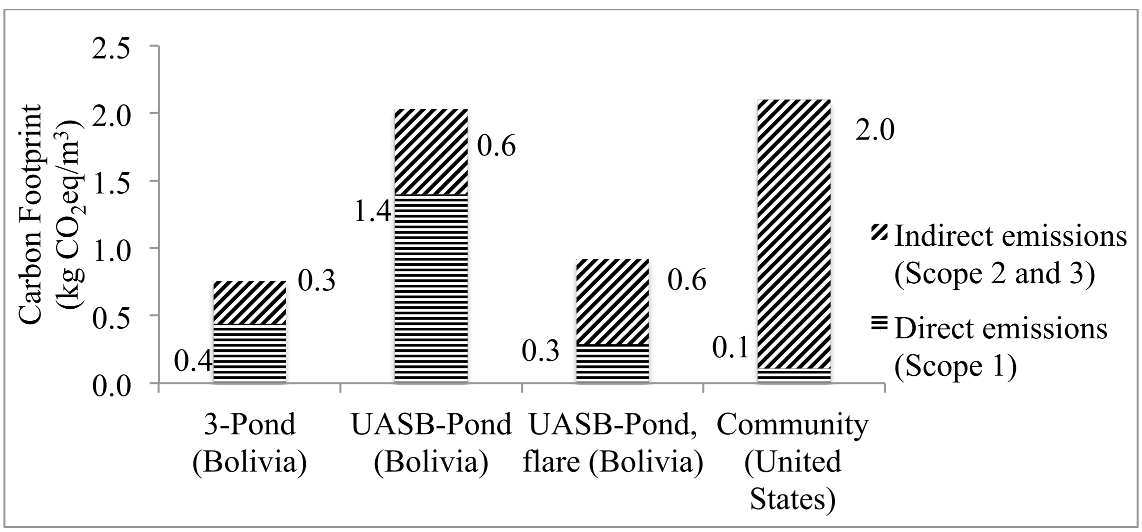

Figure 22. Direct Emissions (Scope 1) and indirect emissions (Scope 2 and 3) contributing to the total carbon footprint of community scale systems in Bolivia and United States

4.7.3 Impact of Context on Eutrophication Potential and Trade-Offs

Context also has an impact of eutrophication potential. The 3-Pond and UASB-Pond system in Bolivia have a higher eutrophication potential than the U.S. community system by a factor of 9.4 and 14.2, respectively (See Figure 23). This can be largely attributed to higher levels of nitrogen and phosphorus in the treated effluent, discharged to nearby surface waters when water is not reclaimed for beneficial reuse. Over 98% of the eutrophication potential from the Bolivia systems comes from direct sources (e.g., nutrients discharged to the environment). In contrast, the U.S community system has a low eutrophication potential because of higher levels of nutrient removal during treatment. In the United States, higher levels of nutrient removal lead to a higher contribution (42%) from indirect sources (e.g., NO_x from electricity) and lower contributions from direct sources (e.g., nutrients discharged), despite a significantly lower eutrophication potential than Bolivia systems under conditions of no water reuse.

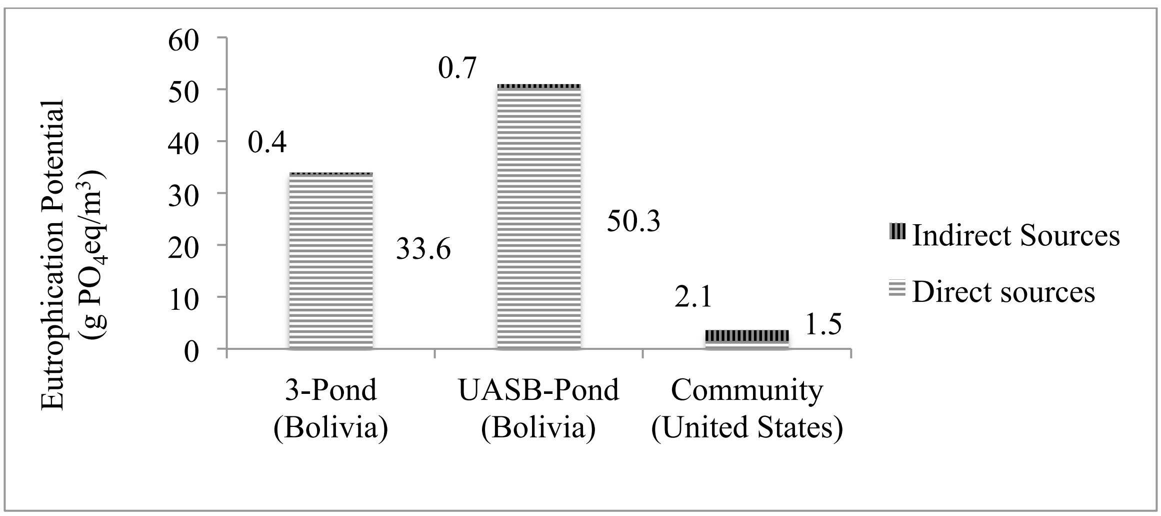

Figure 23. Indirect and direct sources of eutrophication potential from Bolivia (3-Pond and UASB-Pond) and United States (U.S community system)

Additionally, trade-offs emerge between embodied energy, carbon footprint and eutrophication potential. The higher levels of embodied energy used for nitrogen removal for the U.S community system increases the carbon footprint, yet decreases the eutrophication potential. This occurs because more energy is used for nitrogen removal and subsequently, effluent water with a lower concentration of nitrogen is discharged to river. In contrast, the Bolivia systems use less embodied energy for treatment, leading to a lower carbon footprint and higher eutrophication potential when water is not reclaimed. This highlights the importance of matching treatment level to end-use application (e.g., reclaiming nutrient rich water for irrigation purposes). Consequently, global impacts (e.g., carbon footprint and embodied energy) can have direct trade-offs with local impacts (e.g., eutrophication potential). Differences are primarily due to variations in appropriate technologies, since technologies implemented are largely context-dependent (e.g., rural developing world versus urban developed world setting).

4.7.4 Impact of Context on Resource Recovery Strategies

Integrated resource recovery is applicable to WWTPs in both settings; however, limited research has been conducted on how context impacts resource recovery strategies. The offset potential of embodied energy, carbon footprint, and eutrophication potential vary with context depending on the resource recovery strategy implemented. Therefore, water reuse, energy recovery, nutrient recycling, and the integration of all three strategies vary with context for the Bolivia and United States systems investigated. A summary of the percent offset potential of resource recovery strategies (e.g., water reuse, energy recovery, nutrient recycling, and integrated resource recovery) and associated impact categories investigated (e.g., embodied energy, carbon footprint, and eutrophication potential) is shown in Table 25.

For embodied energy and carbon footprint, the offset potential of water reuse for the U.S community system is greater than the offset potential of water reuse in Bolivia. This occurs, because water reuse is more valuable when replacing higher quality water (Shehabi et al., 2012; Tong et al., 2013). In U.S, water reuse is replacing potable water used for non-potable irrigation purposes, whereas in Bolivia water reuse replaces river water used for irrigation. Because the production of potable water has a high embodied energy and carbon footprint, the offset potential of potable water replacement through water reuse is high (15% of the total embodied energy and

18% of the total carbon footprint). In contrast, replacing river water used for irrigation in Bolivia has a minimal impact on embodied energy and carbon footprint offsets (e.g., percent offset of embodied energy offset is 0.2-2.3% for 3-Pond system and 0.1-1.3% for UASB-Pond).

Table 25. Percent offset potential of water reuse, energy recovery, nutrient recycling and integrated resource recovery for embodied energy, carbon footprint and eutrophication potential

Impact Category	Resource Recovery Strategy	3-Pond	UASB-Pond	U.S Community
Embodied Energy (% of total)	Water Reuse	0.2-2.3%	0.1-1.3%	15%
	Energy Recovery	-	18.2%	-
	Nutrient Recycling	-[a]	-[a]	1%
	Integrated Resource Recovery	0.2-2.3%	18.3-19.6%	16%
Carbon Footprint (% of total)	Water Reuse	0.2-2.9%	0.1-0.9%	18%
	Energy Recovery	-	56.7%	-
	Nutrient Recycling	-[a]	-[a]	0.4%
	Integrated Resource Recovery	0.2-2.9%	56.7-57.5%	18%
Eutrophication Potential (% of total)	Water Reuse	19.8-79.2%	19.7-79.0%	16%
	Energy Recovery	-	0.03%	-
	Nutrient Recycling	-[a]	-[a]	6%
	Integrated Resource Recovery	19.8-79.2%	19.7-79.0%	22%

[a]Nutrient recycling offsets accounted for in water reuse offsets, as nutrient benefits in reclaimed water that reduce irrigation needs. Nutrient benefits associated with biosolids in Bolivia are not considered, because biosolids aren't treated and may be considered a hazard to human health

Despite its low impact on embodied energy and carbon footprint in Bolivia, water reuse has a high impact eutrophication potential. In Bolivia, nutrient recycling offsets are included in the water reuse offsets because there is a nutrient benefit associated with water reuse. Consequently, under maximum water reuse (80% or water reclaimed) and crop growth conditions, around 79% of the eutrophication potential can be mitigated for the Bolivia systems. This occurs because nutrients that would otherwise be discharged to the river are diverted for agricultural irrigation through water reuse. In contrast, water reuse in the U.S. leads to a low offset of eutrophication potential (16%) when reclaiming 77% of the water (current practice) from high levels of nitrogen removal and a low fertilizer replacement potential. In the United States eutrophication potential offsets come primarily from the indirect mitigation of NO_x

emissions from electricity avoided through potable water replacement. Therefore, a major difference between developing and developed world technologies is that direct sources of eutrophication (e.g., nutrient not discharged to surface water due to water reuse) are responsible for offsetting eutrophication in the developing world, whereas indirect sources of eutrophication (e.g., electricity avoided through potable water replacement) are primarily responsible for offsetting eutrophication in the developed world.

Energy recovery is only applicable to the UASB-Pond system in Bolivia. This resource recovery strategy is the dominant contributor to offsets for embodied energy (18.2%) and carbon footprint (56.7%). This differs from the U.S community scale system, where aerobic digestion is used instead of anaerobic digestion and energy recovery is not applicable at this scale. It is important to note that energy recovery is not currently practiced at the UASB-Pond system, yet there is a high potential for embodied energy and carbon footprint offset if implemented. Energy recovery's offset potential for eutrophication is low (0.03%) at the UASB-Pond site, because energy recovery has a negligible impact on nutrients discharged to the environment.

Nutrient recycling has low impact on carbon footprint and embodied energy for all systems, but a high impact on eutrophication potential in Bolivia. Nutrient recycling from land application of biosolids is assumed to be only applicable to a U.S. context, because in Bolivia biosolids are not treated and may pose a greater human health risk. In the United States and Bolivia, less than 3% of the carbon footprint and embodied energy is offset from nutrient recycling. However, the eutrophication potential offset associated with the nutrient benefit of water reuse in Bolivia ranges from 20-79%. This wide range depends on the amount of water reclaimed (20-80%) and variations in potential crop yield increase (10-30%). Despite these variations, even under minimal conditions, nutrient recycling in Bolivia through water reuse only has a higher eutrophication potential offset (~20%) than nutrient recycling from water reuse and biosolids land application in United States (6%). Eutrophication potential offsets in U.S. are low compared to Bolivia, because high levels of nutrient removal lead to low fertilizer offset potentials associated with nutrient recycling.

4.7.5 Conclusions for the Impact of Context on WWTPs with Integrated Resource Recovery

Integrated resource recovery leads to the greatest potential benefits in both settings, where the maximum offset potential accounts for water reuse, nutrient recycling, and energy recovery combined with wastewater treatment, as shown in Table 26. This table highlights that the U.S community system has the greatest embodied energy offset potential under integrated resource recovery conditions (5.7 MJ/m^3) primarily due to water reuse, compared to WWTPs with integrated resource recovery offsets in Bolivia (0.28-3.4 MJ/m^3). Despite having the highest integrated resource recovery offset potential, the embodied energy of the WWTP with integrated resource recovery offsets in U.S (30.2 MJ/m^3) is still higher than the systems in Bolivia (12.2-13.9 MJ/m^3). Consequently, the comparatively higher embodied energy offset

Table 26. Summary of embodied energy, carbon footprint and eutrophication potential findings for Bolivia (3-Pond and UASB-Pond) and United States (U.S. community) and maximum offset potential associated with integrated resource recovery

Total with Offset	Units	3-Pond	UASB-Pond	U.S Community
Embodied Energy (MJ/m^3)	Total	12.47	17.29	36.1
	Max. Offset	-0.28	-3.4	-5.7
	Total w/ Offset	12.2	13.9	30.3
Carbon Footprint (kg CO_2eq/m^3)	Total	0.76	2.0	2.1
	Max. Offset	-0.021	-1.2	-0.4
	Total w/ Offset	0.74	0.86	1.7
Eutrophication Potential (g PO_4eq/m^3)	Total	34	51	3.65
	Max. Offset	-27.3	-40.5	-0.8
	Total w/ Offset	7	11	2.8

associated with integrated resource recovery in United States is not large enough to overcome the high embodied energy associated with energy-intensive wastewater treatment technologies.

Similar to embodied energy, the total carbon footprint of the WWTP with integrated resource recovery offsets in United States (1.7 kg CO_2eq/m^3) is higher than the systems Bolivia (0.74-0.86 kg CO_2eq/m^3). The maximum integrated resource recovery offset potential occurs at the UASB-Pond site, primarily due to energy recovery. This highlights that systems integrated natural treatment processes in rural Bolivia have a lower carbon footprint than mechanized systems in an urban U.S. context when considering WWTPs integrated with resource recovery alternatives. Furthermore, it highlights that energy recovery from a community system in Bolivia is more effective at carbon footprint mitigation, than water reuse and nutrient recycling combined in the United States for systems of comparable scale.

Finally, the total eutrophication potential of WWTPs with integrated resource recovery was lowest for the U.S. community system, despite a high maximum eutrophication potential offset associated with water reuse from systems in Bolivia. Significant reductions in eutrophication potential can be achieved through water reuse of nutrient-rich effluents for agricultural irrigation in Bolivia (offsetting 27.3-40.5 g PO_4eq/m^3). Despite this high offset potential, nitrogen removal through energy-intensive nitrification/denitrification processes at the U.S. community system is a more effective way to achieve low eutrophication potential than water reuse at the Bolivia sites. This also highlights trade-offs between global concerns (e.g., carbon footprint, embodied energy) and local concerns (e.g., eutrophication potential), where lower nutrient pollution can be achieved at the expense of higher energy usage and carbon impacts.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

5.1 Scope of Research

This chapter[3] summarizes the major findings of this research by addressing research questions and the stated hypothesis. The following sections discuss key conclusions, limitations, and future work for the framework development (Chapter 2), scale assessment (Chapter 3), and context assessment (Chapter 4). The central hypothesis guiding this research is that: Context and scale impact the environmental sustainability of integrated resource recovery systems applied to the management of wastewater. Three tasks were conducted to answer the following research questions and test the stated hypothesis.

The framework development (Chapter 2) developed a life cycle assessment (LCA) framework for wastewater treatment plants (WWTPs) integrated with resource recovery (water reuse, energy recovery, and nutrient recycling) to answer the following research questions:

- What should be included in the system boundary and what phases should be considered for wastewater treatment and resource recovery systems?
- What input data and emission sources should be considered for these systems?
- What are the main environmental impact categories associated with these systems?
- What should be included in an LCA framework that can assure consistency, and robustness?

[3] Portions (Section 5.5.1) of this chapter are adapted from *Journal of Water Reuse and Desalination* volume *04*, issue number *4*, pages *238-252*, with permission from the copyright holders, IWA Publishing.

- What methods should be used to assess the offset potential of resource recovery?
- What are the major impacting factors of these systems?
- Are certain methods more appropriate to use in certain contexts (developing versus developed world)?

The scale assessment (Chapter 3) investigated the impact of scale on the environmental sustainability of resource recovery systems integrated with wastewater treatment at a household, community, and city scale in a Florida, U.S. context to answer the following research questions:

- How does scale impact technology selection and resource recovery solutions in a developed world settings?
- How does scale impact the environmental sustainability of resource recovery for major impact categories selected (e.g., carbon footprint, embodied energy, and eutrophication potential)?
    - How does scale lead to embodied energy differences between direct and indirect energy (or construction and operation phase)?
    - How does scale lead to carbon footprint differences between direct and indirect emissions (or construction and operation phase)?
    - How does scale impact eutrophication differences between direct and indirect sources of eutrophication potential?
- How do resource recovery strategies mitigate the impact wastewater treatment management at different scales?

The context assessment (Chapter 4) evaluated the impact of context on the environmental sustainability of WWTPs integrated with resource recovery systems for community scale systems in Bolivia and United States to answer the following research questions:

- How does context impact technology selection and resource recovery in developed and developing world settings?
- How does context impact the environmental sustainability of resource recovery for major impact categories selected (e.g., carbon footprint, embodied energy, and eutrophication potential)?
    - How does context lead to embodied energy differences between direct and indirect energy (or construction and operation phase)?
    - How does context lead to carbon footprint differences between direct and indirect emissions?
    - How does context impact eutrophication between direct and indirect sources of eutrophication potential?
- How does context impact the environmental sustainability of resource recovery?
- What knowledge can be transferred to improve sustainability of systems in both settings?

5.2 Framework Development Summary

To develop a comprehensive framework for this research, Chapter 2 reviews existing literature and models on the environmental sustainability of WWTPs integrated with resource recovery. Research gaps, trends, and limitations were identified to develop a robust framework that can evaluate the global and local impacts of context and scale on wastewater management solutions and resource recovery strategies. System boundaries, phases considered, input data required, key environmental impact categories, and varying methodologies appropriate for different contexts were explored.

A review of previous literature determined that comparisons of life cycle impact results from different studies were difficult due to variations in system boundaries, phases considered, parameters considered (e.g., materials, electricity, electricity mix, greenhouse gas emissions (GHGs), chemicals), methodologies used and the presentation of results. The wastewater-energy sustainability tool (WWEST) (Stokes and Horvath, 2010, 2011a) was identified as one of the most sophisticated tools with a comprehensive system boundary for life cycle analysis of wastewater treatment systems. Consequently, WWEST played a central role in aiding the selection of parameters considered, input data collected and the development of a comprehensive framework. Drawing from the various environmental sustainability tools reviewed the following life stages, phases, and parameters were included in the framework:

- Life stages: Construction and operation and maintenance (O&M). Decommission excluded due to a low contribution of less than 1% of the environmental impact (Friedrich, 2002)
- Phases considered: Collection, treatment and distribution
- Parameters considered: Material production and delivery, equipment operation, energy production, sludge disposal and direct emissions (CO_2, CH_4, N_2O), total nitrogen and total phosphorus discharged to the environment
- Resource recovery offsets considered: Energy offsets as natural gas avoided associated with energy recovery, fertilizer offsets associated with nutrient recycling from biosolids

Whereas the WWEST framework contained the most comprehensive set of life stages phases and parameters, certain items were not included in this system boundary. For example, the WWEST framework does not include the mitigation potential of water and nutrients from reclaimed water. Consequently, enhancements were made to the WWEST framework to include a water reuse module to capture water and nutrient offsets associated with water reuse including:

- Water reuse as a co-product to replace water with varying end-uses (e.g., water reuse to replace river water for irrigation, water reuse to replace potable water for irrigation).
- Nutrient benefit of reclaimed water used for irrigation for varying end-uses (e.g., nutrient benefit of replacing river water irrigation with reclaimed water, nutrient benefit of replacing synthetic fertilizers through water reuse).

A process-based life cycle assessment (LCA) approach was used for analysis in the current research. Process-based LCA was selected because of its flexibility and applicability to different settings (developing and developed world). Additionally, process-based LCA allows for the analysis of specific unit processes and the separation of results by unit processes. International Organization of Standardization (ISO) 14040 guidelines were followed for analysis (ISO, 2006) by defining the scope and goal of the research, conducting a life cycle inventory analysis, conducting a life cycle assessment, and interpreting results. A functional unit of 1 cubic meter of treated wastewater over a 20-year life cycle was selected. Life cycle inventories were collected through site visits to facilities and interactions with engineering practitioners. The life cycle assessment was conducted to evaluate the environmental impact of WWTPs with integrated resource recovery in Bolivia and the United States through case studies using SimaPro 7.2 and SimaPro 8, PhD version. Subsequently, sensitivity and uncertainty analyses were conducted, because uncertainty can emerge due to variations in input parameter ranges (e.g., seasonal variations in nutrient discharges, seasonal fluctuations in electricity usage).

Embodied energy, carbon footprint, and eutrophication potential were identified as key environmental impact categories used to assess environmental sustainability of WWTPs integrated with resource recovery. Consequently, these categories were selected to evaluate global impacts (e.g., embodied energy, carbon footprint) and local impacts (e.g., eutrophication potential) of the water-energy-carbon-nutrient nexus as it applies to wastewater management solutions and resource recovery strategies. To identify critical mitigation areas and enable accurate comparisons, impact categories were separated by direct and indirect emission sources. Embodied energy represents the life cycle energy consumption including direct energy (e.g., electricity production) and indirect energy (e.g., production of materials, chemicals). Carbon footprint represents the life cycle greenhouse gas emissions including direct (Scope 1) emissions (e.g., direct CO_2, CH_4, and N_2O), indirect (Scope 2) emissions (e.g., electricity production), and other indirect (Scope 3) emissions (e.g., production of materials and chemicals). Lastly, eutrophication potential represents nutrient pollution over the life cycle including direct sources (e.g., nutrients discharged directly to the environment) and indirect sources (e.g., NO_x emissions from electricity, material, and chemical production). Contributions from collection, treatment, distribution and resource recovery offset potentials were investigated over construction and operation and maintenance phases. Through a thorough review of previous literature and models, this chapter developed a comprehensive life cycle framework to evaluate scale and context's influence on the environmental sustainability of WWTPs integrating resource recovery.

5.3 Scale Assessment Summary

Using the framework developed in Chapter 2, scale's influence on the environmental sustainability of WWTPs integrating resource recovery was evaluated in Chapter 3. Systems designed for treatment and reuse were evaluated at the household, community, and city scale in Tampa, FL, a coastal city facing urbanization and population growth. The systems selected were mechanized technologies appropriate and applicable to an urban developed world setting. These systems were designed to meet stringent water quality standards in a densely populated urban U.S. city facing effective nutrient management needs and vulnerability to climate change.

The three systems analyzed include: (1) Household (250 gpd) septic tank followed by an aerobic treatment unit (ATU) serving 1 home (2-3 people) with subsurface landscape drip irrigation reuse, (2) Community (0.3 mgd) advanced water reclamation facility with nitrification/denitrification using headworks (grit removal, bar screens, odor scrubbing), equalization tanks, aeration tanks, denitrification tanks, re-aeration, clarifiers, denitrification filters, clearwell, chlorination and UV disinfection, aerobic digestion serving approximately 1,500 population equivalents (p.e.) with golf course irrigation reuse and some surface water discharge (3) a city scale (10.3 mgd) advanced water reclamation facility with headworks (grit removal, bar screens), activated sludge (biological secondary treatment including aeration and return activated sludge), secondary clarification, filtration, chlorination, anaerobic digestion for energy recovery serving approximately 100,000 p.e. with residential landscape irrigation reuse and some deep well injection to prevent salt water intrusion.

This research found that global impacts (e.g., embodied energy and carbon footprint) adhere to economies of scale, where larger systems have lower impacts despite fluctuations in relative contributions from varying parameters. Water reuse distribution has a lower impact than treatment compared to other regions (e.g., California) due to differences in topographical conditions. In this study, Florida's flat topography appears to favor centralization of wastewater management (around 10 mgd) over smaller decentralized and semi-centralized systems, particularly when energy-efficient variable frequency drive pumps are used for water reuse distribution at the city scale. Beyond Florida, other regions worldwide characterized by flat topographies may favor centralization at 10 mgd as a viable wastewater management solution.

Household systems had the largest impact in embodied energy, carbon footprint and eutrophication potential, where electricity usage for treatment and distribution, methane emissions from the septic tank, and higher levels of nutrient discharged to the environment were key contributors to the environmental impact categories evaluated. Consequently, the life cycle impacts of less energy-intensive passive nutrient reduction techniques with gravity trenches designed to maximize water and nutrient reuse potential merit further investigation.

At the community scale, high energy usage during treatment led to a higher embodied energy and carbon footprint for treatment, but a lower eutrophication potential due to more advanced nutrient removal. This highlights a key trade-off between global (e.g., embodied energy and carbon footprint) and local impacts (e.g., eutrophication potential), where advanced treatment for nutrient removal effectively reduces nutrient pollution at the expense of higher energy usage and greenhouse gas emissions. Additionally, higher electricity usage leads to a higher relative contribution of indirect sources of eutrophication (e.g., NO_x emissions from electricity), compared to direct sources of eutrophication (e.g., nutrients discharged to the environment). This research suggests that mitigation of global impacts could be achieved by matching treatment level to end-use application by accommodating for seasonal fluctuations. For example, high levels of nitrogen removal may only be needed when discharging to surface water bodies during the rainy season, whereas less stringent nitrogen regulations could be put in place when reclaiming water for beneficial irrigation during the dry season.

The city scale achieved the lowest carbon footprint and embodied energy due to economies of scale. This occurs despite the increase in relative contribution from piping infrastructure, chemicals, and direct N_2O emissions from biosolids. A dominant factor in reducing the embodied energy and carbon footprint at the city scale is the decrease in electricity consumption per cubic meter compared to decentralized (household) and semi-centralized (community) scale alternatives. On the other hand, the city scale has a larger eutrophication potential than the community scale because nitrogen removal is lower and there is an increased contribution in nitrogen discharged to soil through both biosolids and reclaimed water at the city scale. Compared to nitrogen discharges, phosphorus discharges contribute less to eutrophication potential at all scales; however, this may be due to assumptions embedded in the eutrophication fate and transport model, where, both nitrogen and phosphorus are limiting.

Scale of implementation and technologies implemented also impact the preferred combination of resource recovery strategies and the associated mitigation potential. Whereas the city scale benefits from integrated resource recovery (e.g., combined water reuse, energy recovery, and nutrient recycling), only water reuse and nutrient recycling are applicable at the household and community scale. Water reuse had the highest mitigation potential of both global and local impact categories at all scales, where potable water offsets are highest at the household level since all of the water is reclaimed at this scale. Nutrient recycling has the lowest mitigation potential for all impact categories, yet fertilizer offsets increase with scale due to a higher production of nutrient-rich biosolids replacing fertilizers at larger scales. The city scale achieves the greatest energy offsets, where the integration of all three forms of resource recovery leads to a 49% offset of embodied energy. This is approximately equal to the embodied energy of treatment and greater than the energy needed for water reuse, highlighting the benefits of integrating water reuse, energy recovery, and nutrient recycling. These findings highlight that there may be benefits hybrid systems, where water is reclaimed at the community scale and biosolids are treated at a centralized facility. This would lead to the beneficial increase of potable water offsets from semi-centralized community scale water reuse, while increasing fertilizer offsets from biosolids and energy offsets from energy recovery at the larger centralized city scale.

5.4 Context Assessment Summary

In Chapter 4, two community scale systems were investigated in rural Bolivia and then compared to the community scale U.S system investigated in Chapter 3. These Bolivian systems integrate natural wastewater treatment technologies appropriate, require less mechanical energy inputs, and applicable to a rural community adjacent to agricultural areas, serving small towns in a developing world context. The U.S. system is a mechanized, energy-intensive technology in an urban area near a golf course, serving a gated community in a developed world context. The community-managed systems in rural Bolivia were compared to the community scale system in urban United States to evaluate the influence of context (e.g., location, treatment technology, resource recovery strategy, demographics) on the environmental sustainability of wastewater management solutions and resource recovery strategies implemented.

Technologies and resource recovery applications vary with context. The systems evaluated in Bolivia include an upflow anaerobic sludge blanket (UASB)-Pond system (UASB reactor followed by maturation ponds) and a 3-Pond system (facultative pond followed by maturation ponds). The U.S community system implements primary, secondary, tertiary disinfection via UV and chlorination, and aerobic digestion. Whereas the UASB-Pond system is the only system at this scale with energy recovery from the UASB reactor, all systems have the potential to practice water reuse and nutrient recycling. In Bolivia, potential agricultural reuse replaces river water irrigation, where reclaimed water has an additional nutrient benefit. In this context, the nutrient benefit from water reuse leads to a reduction in electricity needed for agricultural irrigation compared to river water irrigation, since less water is needed for a comparable crop yield. No fertilizers are replaced in Bolivia since crops are grown organically and nutrient recycling from biosolids are not considered due to the low frequency of sludge removal at the 3-Pond site and potential hazards associated with reclaiming untreated biosolids. In contrast, water reuse at the U.S. community system replaces potable water and both reclaimed water and biosolids have a nutrient benefit. Fertilizers offset through nutrient recycling are considered in the U.S. context, since synthetic fertilizers are currently used in this region.

Both the total embodied energy and carbon footprint in Bolivia were lower than the U.S. community system, primarily due to lower operational electricity requirements associated with natural system integration compared to mechanized systems. Despite having a lower embodied energy associated with treatment, the embodied energy of collection for the Bolivia systems had higher contribution since less densely populated rural regions can lead to higher infrastructure requirements for collection compared to densely populated urban regions in United States where less land is available and treatment occurs closer to the population served. For carbon footprint, direct (Scope 1) emissions from treatment processes (CH_4 from UASB reactor and ponds) were dominant contributors to the Bolivia systems, whereas indirect (Scope 2) emissions (e.g., electricity) were the dominant contributor to the U.S. community system. Consequently, carbon footprint mitigation efforts in rural developing regions should focus on energy recovery efforts from anaerobic treatment processes, flaring (current practice) or the implementation of systems without anaerobic treatment processes (e.g., 3-Pond system). In contrast, carbon and energy mitigation efforts of mechanized systems in urban developed regions should focus on reducing electricity consumption (e.g., variable frequency drive pumps, energy-efficient aeration, waste heat recovery).

When evaluating eutrophication potential, trade-offs emerge between global impacts (e.g., embodied energy and carbon footprint) and local impacts (e.g., eutrophication potential). Whereas the two Bolivia systems benefit from a lower embodied energy and carbon footprint, they have a higher eutrophication potential, largely due to higher levels of nitrogen and phosphorus in the treated effluent directly discharged to surface waters. In United States, the community scale system achieves lower eutrophication potential due to more energy-intensive mechanized treatment implemented to reduce nutrient loads. This leads to a higher contribution from indirect sources of eutrophication potential in the U.S., but a significantly lower eutrophication potential due to lower levels of direct sources of eutrophication potential. These differences emerge due to changes in treatment technologies, which are largely context-dependent; highlighting context's impact on the environmental sustainability of wastewater treatment systems in a rural developing world and urban developed world setting.

Resource recovery strategies and associated offsets also shift with context. For example, the embodied energy and carbon footprint offset potential of water reuse in the United States is greater than the offset potential of water reuse in Bolivia. This occurs because replacing higher quality water (e.g. potable water) leads to greater energy savings than replacing lower quality water (e.g., river water). In contrast, eutrophication potential offsets of water reuse are higher in Bolivia, since nutrient benefit associated with water reuse increases as more water is reclaimed and direct surface water discharges of nutrients are avoided. This highlights the importance of matching treatment level to end-use application, especially in developing world regions where energy-intensive advanced treatment for nutrient reduction is less appropriate. Energy recovery from the UASB-Pond systems is the dominant contributor to carbon and energy offsets in Bolivia. This differs from the U.S. community system, where energy recovery is not applicable based on technology selection (e.g., use of aerobic digestion). Accounting for all integrated resource recovery offsets, the Bolivia systems have lower global impacts (e.g., embodied energy and carbon footprint) and the U.S. community system has lower local impacts (e.g., eutrophication potential). In addition Bolivia's UASB-Pond system highlights the benefits of combining waste-to-energy systems with natural treatment processes for water reuse. Additionally, research on both developing and developed world applications leads to an increase in international knowledge transfer, which can provide sustainable and appropriate solutions to wastewater management and resource recovery in both settings.

5.5 Limitations and Future Work

5.5.1 Framework Development Limitations and Future Work

Several key attributes were identified from the environmental sustainability tools reviewed in this research that would be beneficial to include in a single robust LCA framework on WWTPs with integrated resource recovery in future works. The key attributes to include in future frameworks are: (1) a user-friendly web-based interface, (2) a dynamic model that captures how GHG emissions respond to operational changes, (3) offset potential associated with a wide range of resource recovery strategies and (4) model calibration and validation (Table 27).

Table 27. Useful attributes from environmental sustainability tools for wastewater that would be beneficial to include in future frameworks

Estimation Tool	Useful Attributes	Benefit of Attribute
CHEApet[a]	User-friendly web-based tool containing some tertiary filtration and UV disinfection estimation capabilities. Future versions will include biological and chemical phosphorus removal, step-feed BNR, and chlorine disinfection estimation abilities, which would be useful to making a more robust tool.	The web-based interface is beneficial to user-friendliness, while process-specific estimation capabilities can increase transferability of technology comparisons.
BSM2G[b]	A dynamic process-based tool that captures variations in operating conditions, temperature, and influent loads over time.	Dynamic modeling desalination unit processes or tertiary treatment processes for water reuse could be beneficial to a robust tool.
GPS-X[c]	Future version of GPS-X will include offsets due to fertilizers and carbon sequestration from land use. Additionally, it can be used to evaluate how process changes affect emissions. The GPS-X model was also tested against carbon footprint data from a wastewater treatment facility to calibrate and validate the accuracy of results.	This is the only tool that used calibration and validation to verify results, which would be useful to the development of a robust water reuse carbon footprint estimation tool.
mCO2[d]	User-friendly software that automatically produces a report identifying critical areas to meet emission criteria.	User-friendly software is a crucial element to the successful development of a carbon footprint tool for water reuse or desalination systems.

Sources: [a]Crawford et al. (2011); [b]Corominas et al. (2012); [c]Goel et al. (2012); [d]MWH (2012).

Some wastewater carbon footprint estimation tools (e.g., carbon heat energy analysis plant evaluation tool (CHEApet) and mCO2) contain user-friendly interfaces. Similar to WESTWeb, CHEApet provides a web-based interface, whereas mCO2 software automatically produces a report to identify critical mitigation areas (Crawford et al., 2011; MWH, 2012). These examples of user-friendly attributes in future models could lead to greater adoption in both research and engineering practice.

A robust estimation tool should also contain dynamic quantifications of how operational changes impact results. To capture the impact of operational changes, the Benchmark Simulation Model Platform No. 2 (BSM2G) includes a dynamic process-based GHG estimation tool that can analyze how changes in the system (e.g., hydraulic load, influent water quality, temperature, operational modifications) impact direct N_2O and CH_4 emissions from secondary treatment (i.e., activated sludge) and sludge processing (i.e., anaerobic digestion) (Corominas et al. 2012). This would be useful to incorporate in a user-friendly LCA analysis tool.

Additionally, accounting for the offsets associated with a wide range of resource recovery practices would also be beneficial to practitioners and researchers. This would allow for comparisons of varying resource recovery strategies, shown in Table 28. The GPS-X tool includes offsets due to the recovery of energy, fertilizers and carbon sequestration from land use (Goel et al. 2012), whereas WWEST includes offsets associated with energy and fertilizer co-products (Stokes and Horvath 2011a). Future research could expand on this work by quantifying the environmental impacts of varying resource recovery strategies applicable for different scales and contexts.

Model validation is also important to ensure the accuracy of results in future studies. For example, carbon footprint estimates from the GPS-X tool were calibrated to match actual data

(Goel et al. 2012). Estimates of direct emissions can be validated through comparisons to GHG emissions monitored on-site. Determining the contribution of specific treatment steps may require energy estimations for each unit process since this data is often not collected in practice. Energy estimation equations have been developed for some water and wastewater unit processes and should be validated using actual energy consumption data (Carlson and Walburger 2007; Johnston, 2011).

Table 28. Different resource recovery strategies for energy recovery, nutrient recycling and water reuse

Resource Recovery Type	Technologies and Applications
Energy Recovery	Combined heat and power Biosolids incineration Effluent hydropower Onsite wind and solar power Heat pump Bioelectrical systems Microalgae
Nutrient Recycling	Biosolids land application Urine separation Struvite crystallization Aqua-species
Water Reuse	Agricultural irrigation Industrial reuse Urban reuse Indirect potable reuse Direct potable reuse

Source: Adapted from Mo and Zhang (2012b)

5.5.2 Scale Assessment Limitations and Future Work

The current research uses a process-based LCA model to evaluate scale's influence on wastewater management solutions and resource recovery strategies through case studies in the developed world. A major limitation of process-based LCA is the data-intensive and time-consuming nature of collecting and analyzing all of the inventory items needed to comprehensively evaluate these systems. This may limit widespread adoption of LCA models, particularly outside of academic settings. Future research should attempt to overcome this challenge by developing a predictive model with minimal inputs, capable of capturing the behavior of important environmental indicators.

This prediction model could be applied to wastewater and resource recovery strategies across different scales to further understand the impact of scale on varying systems with varying end-uses for water, energy, and nutrient reutilization. By combining environmental input-output life cycle assessment (EIO-LCA), economies of scale equations for wastewater, and offset costs associated with resource recovery strategies, a model can be developed using minimal input data to estimate the impact of scale. The cost of each system can be estimated utilizing existing economies of scale equations (EPA, 1978a, 1978b; Fraquelli and Giandrone, 2003; Hopkins et al., 2004; Walski, 2012), in which system size is the only input required. Both construction cost and operation cost can be calculated separately for the existing wastewater treatment systems at each scale. Subsequently, these costs would serve as inputs to calculate the embodied energy and carbon footprint of each system using EIO-LCA. The percent contribution from construction and operation phases can be compared to process-based case study results to evaluate the accuracy of the prediction model. If the behavior of the prediction model is comparable to the process-based case studies, economies of scale can be evaluated in terms of environmental impact using system size as the sole input parameter.

The cost of resource recovery alternatives should also be considered to determine the mitigation potential of resource recovery. For example, if water reclamation replaces potable water, the cost of potable water production should be used as the input to the EIO-LCA method to determine the energy and carbon offset of water reuse. If the predicted model can estimate the impact of wastewater treatment and resource recovery alternatives using system size and cost data as the only inputs, this research can provide a useful tool to evaluate the impact of scale in a simple, yet robust model. The predictive model would not be applied to systems in the developing world context, because EIO-LCA methods do not contain economic input output tables appropriate for Latin American countries. Therefore, this model may not be regionally transferable.

Another limitation of the current study lies in the limited number of case studies investigated and the limited technologies selected for investigation. Conclusions are only based on three systems, where both technology change and scale were found to impact the environmental sustainability of WWTPs with resource recovery. Future research could investigate technology change and scale individually. For example, it would be beneficial to evaluate scale's influence on environmental sustainability for the same technology implemented at various scales, increasing the number of case studies for a wider range of wastewater treatment capacities. Five to ten systems could be selected within the range of completely decentralized household systems to larger centralized systems (greater than or equal to 100 mgd), with no changes in technology. This could then be compared five to ten systems implementing a different technology at the same scales. This would allow for a comparison of different technologies at the same scale and the same technology at different scales. Enough systems would need to be selected to make the results statistically significant, in order to produce a regression model to estimate environmental impact for a given technology. It would be useful to investigate both proven and emerging technologies with innovative resource recovery strategies. For example, nitrogen recovery strategies could be compared to phosphorus recovery strategies at different scales, as well as the integration of nitrogen and phosphorus recovery. Beyond resource recovery, other strategies for energy reductions in the water and wastewater sector could be investigated (e.g., demand-management strategies, energy-efficient appliances, grey water reuse) to identify what combination of technologies and energy reduction strategies can move water and wastewater management towards carbon neutrality and effective nutrient management. This research would be beneficial to researchers, practitioners, and decision-makers, leading to potentially transformational thinking on management of water and wastewater.

5.5.3 Context Assessment Limitations and Future Work

The context assessment consisted of a comparison between systems applicable to a rural developing world setting (e.g., Bolivia) and an urban developed world setting (e.g., United States). Only two case studies were conducted in Bolivia and these case studies were compared to only one community scale system in the United States. It would be useful to compare other technologies at different scales in these settings to see if trends between rural and developing settings change with scale and technology. For example, future research could compare household, community, and city scale systems in both settings, whereas the current research was limited to only community scale comparisons. At the household scale, for example, decentralized household wastewater treatment solutions integrated with resource recovery in both settings could be compared expanding comparisons to systems in other settings as well. It would be interesting to compare composting latrines in South America to septic systems in the United States and on-site source separation technologies in Europe. The same could be done for city scale systems for commonly used technologies in different regions. Other contexts also merit further investigation. For example, technologies applicable to urban developing world settings and rural developed world settings could be compared. The goal of context comparisons should be to obtain useful information that leads to international knowledge transfer to improve energy, carbon, and nutrient management in both settings.

Other impact categories could also be evaluated. The current study focused on embodied energy, carbon footprint and eutrophication potential to evaluate the water-energy-carbon-nutrient nexus as it relates to wastewater management and resource recovery. However, LCA tools can be used to investigate a wide range of environmental impact categories (e.g., carcinogens (chloroethylene [C_2H_3Cl] equivalents), ozone depletion (CFC-11 equivalents), respiratory organics (ethylene [C_2H_4] equivalents), aquatic ecotoxicity (triethylene glycol [TEG] water), terrestrial ecotoxicity (TEG soil)). Other categories could be investigated to identify comprehensive impacts of systems over their life cycle.

Additionally, the current study assumes that treatment technologies in both regions treat wastewater to a suitable standard for safe reuse. In reality, pathogens may be a more pressing issue in rural Bolivia, whereas emerging contaminants and personal care products may be more of a concern in an urban U.S. context. Consequently, other important environmental impact categories emerge depending on technology and context. Further research is needed to evaluate how a wide range of environmental impact categories impact other global and local concerns for environmental sustainability.

Analysis could also be done with varying methodologies within an LCA framework to investigate how results change with different methods investigated. For example, different methodologies to assess eutrophication potential are available in SimaPro 8 (PhD version). Future research could compare different methodologies to analyze how changes in methodology shift the eutrophication potential results. Additionally, results modeled in life cycle assessment software could be compared to on-site measurements of eutrophication potential to test the accuracy of eutrophication potential modeling in different regions.

Lastly, current research could be expanded to explore context and scale's impact on a broader scope of sustainability. This broader scope of sustainability would integrate social, economic, and environmental factors related to effective wastewater management and resource recovery solutions. LCA can be used to investigate environmental impacts and life cycle cost (LCC) analysis can be used to assess economic impacts. Social impacts are related to both technical and non-technical factors (e.g., regulations, local preferences, location, funding sources available, operation and maintenance requirements, and population demographics) that lead to different wastewater management and resource recovery practices. These factors can impact technology selection, social acceptance of resource recovery strategies, and differences in practice related to water, energy, and nutrient reclamation. For example, in the U.S. context, it is assumed that residents irrigating with nutrient rich reclaimed water use less fertilizer; however, further research is needed to determine how the use of reclaimed water impacts fertilizer usage. In some cases, residents may not be aware that they are using reclaimed water, highlighting the need to educate the public about the benefits of resource recovery. Research is needed to determine what technical and non-technical factors have a major impact of these social factors, which can impact the environmental sustainability systems at varying scales and in varying contexts.

Trade-offs are expected to emerge between environmental, economic, and social factors; however, this information can be used to design a decision-making tool for scale appropriate, socially acceptable, environmentally sustainable and economically feasible wastewater management solutions and resource recovery strategies for communities. Understanding the complexities of decision-making as it relates to wastewater management and resource recovery strategies is crucial to moving towards sustainable solutions as they pertain to cost-efficient, scale and context appropriate solutions for the water-energy-carbon-nutrient nexus.

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APPENDICES

Appendix A. Data Collection, Calculations, and Life Cycle Inventory for United States

A.1 Infrastructure

Data related to pipe diameter, pipe material, and pipe length was collected for wastewater collection and water reuse distribution for household, community, and city scale systems. Subsequently, pipe mass was calculated based on assumed pipe densities from various manufacturers (i.e., U.S. Plastics, Cooper Industries, Peterson Products, etc.). Collection piping for the household system was assumed to be negligible due to the short distance needed to transport wastewater to an on-site septic tank. Additionally, data on tank sizes and tank material were collected to estimate volumes of reinforcing steel and concrete in treatment tanks for each system. Reinforced steel was assumed to be 2% of the concrete volume, similar to water energy sustainability tool (WESTWeb, 2015). Cost data on pumps, valves and fittings were collected as well, though this data was only available at the household scale and had a negligible effect on the environmental impact. Diesel consumption for excavation was assumed to have a negligible impact, because the operation and maintenance phase is the dominant contributor to the environmental impact over the life cycle. Material delivery was assumed to have a negligible impact over the life cycle, since most materials can be produced within the State of Florida.

A.2 Operation and Maintenance

Electricity data was collected from the WERF decentralized cost estimation tool (WERF, 2010) at the household scale and directly from WWTP operators at the community and city scale. Annual electricity usage was collected for the household and community scale systems, whereas monthly electricity usage data was available at the city scale. Annual chemical usage data was collected for the city and community scale, whereas chemicals were not used at the household scale. Sludge removal electricity and transport per cubic meter of wastewater treated were assumed to be the same at all scales, whereas operational diesel consumption per cubic meter of wastewater treated for treatment and distribution was assumed to be the same at the community and city scales.

Direct CH_4 emissions from anaerobic treatment processes (e.g., septic tank at household scale, anaerobic digester at city scale), N_2O emissions from nitrification processes, and N_2O from biosolids land application were estimated using EPA and IPCC methods (IPCC, 2006; EPA 2010). Biogenic CO_2 emissions were also calculated, but these emissions are considered negligible by the IPCC (IPCC, 2006).

Table A1. Life cycle inventory for construction of WWTPs with integrated resource recovery at different scales. Inventory items expressed per cubic meter of treated water

Stage	Item	Household	Community	City
Collection	Piping - PVC (kg/m^3)		0.015 (0.007-0.018)	0.011
	Piping - VCP (kg/m^3)			0.188
	Piping - Concrete (m^3/m^3)			0.000
	Piping - Reinforcing steel (kg/m^3)			0.013
	Piping - HDPE (kg/m^3)			0.002
Treatment	Tanks - Concrete (m^3/m^3)	0.0009 (0.0007-0.0012)	0.00014 (0.00012-0.00016)	0.00008 (0.00007-0.00010)
	Tanks - Reinforcing steel (kg/m^3)	0.15 (0.11-0.19)	0.022 (0.018-0.026)	0.013 (0.011-0.016)
	Excavation - Diesel (kg/m^3)	0.009 (0.005-0.014)		
Distribution	Piping - PVC (kg/m^3)	0.0001	0.002	0.005
	Piping - Cast Iron (kg/m^3)			0.188
	Piping - Ductile Iron (kg/m^3)			0.000
	Piping - Galvanized steel (kg/m^3)			0.013
	Piping - Steel (kg/m^3)			0.011
	Piping - Concrete (m^3/m^3)			0.002
	Piping - Reinforcing Steel (kg/m^3)			0.000
	Pump Tank, Concrete (m^3/m^3)	0.0003		
	Reinforcing steel (kg/m^3)	0.0485 (0.0476-0.0494)		
	Pump, 12 gpm (2009USD/m^3)	0.035 (0.032-0.037)		
	Valves (2009USD/m^3)	0.031 (0.029-0.034)		
	Plastic pipe fittings (2009USD/m^3)	0.015 (0.014-0.016)		
	Other fittings (2009USD/m^3)	0.013		

Equations to calculate biogenic CH_4 are shown in Table B1 (See Appendix B), and the equation to calculate N_2O emissions from WWTPs is shown below:

$$N_2O_{wwtp} = Q*TKN*EF_{N2O}*(44/28)*1E\text{-}03 \quad (1)$$

N_2O_{wwtp} is the N_2O emissions generated from WWTP process (kg N2O/yr) and Q is the wastewater influent flow rate (m^3/year). This equation was modified to calculate emissions per year. This equation also includes the influent TKN (mg/L), the N_2O emission factor, EF_{N2O} (0.005 g N emitted as N_2O per g TKN) (Chandran, 2010), and a conversion factor modified to calculate kg N_2O/year. The N_2O from land applied biosolids was calculated using the following equation:

$$N_2O_{biosolids} = (44/28)*F_{on}*EF_1 \quad (2)$$

$N_2O_{biosolids}$ is the nitrous oxide generated from land applied biosolids, where F_{ON} is the annual amount of biosolids or other additions of nitrogen applied to soils (kg N/year) and EF_1 is an emission factor for nitrogen additions from organic amendments as a result of the loss of soil carbon (kg N_2O-N/kg N). High uncertainty is associated with EF_1, where this value ranges from 0.003-0.03 (IPCC, 2006). The amount of nitrogen in biosolids was calculated by collecting the amount of biosolids hauled per year and the percent total nitrogen within the biosolids. At the city scale data on the percent of total nitrogen in biosolids was collected directly from the facility. At the household and community scale, this data was not available so a range of typical values from previous literature was used (Tchobanoglous et al., 2004)

Nutrient discharges to the environment were collected at each scale. Nitrogen and phosphorus discharges from surface water and reclaimed water to soils were collected. Additionally, nitrogen and phosphorus discharges to soil from biosolids were collected. This

Table A2. Life cycle inventory of operation and maintenance of WWTPs with integrated resource recovery at different scales. Inventory items expressed per cubic meter of treated water

Stage	Item	Household	Community	City
Collection	Electricity (kWh/m^3)		0.04 (0.001-0.26)	0.07 (0.03-0.12)
Treatment	Caustic Soda (kg/m^3)			0.002
	Sodium hypochlorite (kg/m^3)			0.21 (0.14-0.27)
	Chlorine (kg/m3)		0.11	
	Ferric sulfate (kg/m^3)		0.0215 (0.0210-0.0219)	
	Methanol (kg/m^3)		0.004	
	Polymer (kg/m^3)		0.009 (0.006-0.012)	
	Electricity (kWh/m^3)	1.11	1.83	0.12 (0.08-0.17)
	Direct CH_4 (kg CH_4eq/m^3)	0.02 (0.002-0.05)	-	0.02 (0.007-0.03)
	Direct N_2O (kg CO_2eq/m^3)	0.16 (0.12-0.21)	0.09 (0.05-0.16)	0.07 (0.06-0.10)
	Direct N_2O - biosolids (kg CO_2eq/m^3)	0.003 (0.001-0.01)	0.01 (0.003-0.05)	0.05 (0.01-0.23)
	Sludge removal electricity (kWh/m^3)	4.5E-05 (1.8E-05-7.2E-05)	4.5E-05 (1.8E-05-7.2E-05)	4.5E-05 (1.8E-05-7.2E-05)
	Sludge removal transport (tkm/m^3)	0.0023 (0.0021-0.0027)	0.0023 (0.0021-0.0027)	0.0023 (0.0021-0.0027)
	Diesel (kg/m^3)		0.016 (0.10-0.43)	0.016 (0.10-0.43)
Distribution	Electricity (kWh/m^3)	1.4	0.5	0.20 (0.10-0.43)
	Diesel (kg/m^3)		0.025 (0.0004-0.28)	0.025 (0.0004-0.28)
Discharges to environment	N to surface water (g/m^3)		0.65 (0.34-4.93)	
	P to surface water (g/m^3)		0.13 (0.02-0.77)	
	N to soil from water reuse (g/m^3)	16.4 (2.0-30.8)	0.2 (0.03-6.8)	2.3 (1.3-3.1)
	P to soil from water reuse (g/m^3)	0.16 (0.12-0.20)	0.005 (0.004-0.04)	0.01 (0.004-0.03)
	N to soil for biosolids (g/m^3)	0.3 (0.04-0.8)	1.3 (0.2-3.0)	4.5 (0.7-12.6)
	P to soil from biosolids (g/m^3)	0.014 (0.008-0.027)	0.06 (0.04-0.10)	0.09 (0.06-0.15)
Resource Recovery	Potable Water Offsets (MJ/m^3)	7.17	5.55	4.03
	N Fertilizer Offsets - water reuse (g/m^3)	30.0 (20.0-40.0)	0.001 (0.0002-0.007)	0.009 (0.004-0.013)
	P Fertilizer Offsets - water reuse (g/m^3)	8.0 (6.0-10.0)	0.0002 (0.0001-0.001)	0.006 (0.0002-0.0014)
	N Fertilizer Offsets - biosolids (g/m^3)	0.65 (0.42-1.03)	3.0 (2.1-3.9)	10.4 (7.2-16.3)
	P Fertilizer Offsets - biosolids (g/m^3)	0.71 (0.39-1.37)	3.2 (1.9-5.1)	4.6 (2.8-7.4)
	Energy Offsets -natural gas (kg/m^3)			0.02 (0.01-0.03)

nutrient data was collected directly from WWTPs and typical values from previous literature were used when data was not available (Tchobanoglous et al., 2004; Asano et al., 2007).

Additionally, data on resource recovery offsets were collected to calculate the beneficial offsets from water reuse, nutrient recycling, and energy recovery. Potable water offsets include chemicals and electricity offset from potable water production in Tampa, FL from a previous study (Santana et al., 2014). Fertilizer offsets assume all of the nutrients discharged in reclaimed water and biosolids replace nitrogenous and phosphorus-based fertilizers. Energy offsets assume methane produced at the community scale replaces natural gas as shown in Table B1.

Appendix B. Data Collection, Calculations, and Life Cycle Inventory for Bolivia

B.1 Infrastructure

Data collected in the field was compiled or calculated to obtain: (1) the mass (kg), area (m^2), or volume (m^3) of materials produced; (2) freight transportation (tkm) of materials delivered; and (3) electricity (kWh) and fuel (kg) of equipment consumed on-site as required by SimaPro 7.2. For material delivery, it was assumed that truck with a 16 ton or greater carrying capacity was used to ship materials.

B.2 Operation and Maintenance

To estimate the electricity use for electrical equipment, the national Bolivian electricity mix (44% fossil fuels, 54% hydropower, and 1.5% other) was used (CIA, 2012). Fuel consumption rates were obtained from manufacturer data (e.g., Caterpillar (1998)), and the WEST tool (Available upon request at west.berkeley.edu/).

Electricity and fuel required for sludge disposal were associated with pumping water out the facultative lagoon, removing the sludge with an excavator, and replacing the geomembrane at the 3-Pond site. The cumulative volume of sludge produced and removal frequency needed upon reaching 25% of the lagoon volume (Oakley, 2006) was calculated using TSS samples (n=4) taken in the field from 2008 to 2011.

The fuel consumption needed to remove this accumulated sludge using a mid-sized excavator (150 HP) was then calculated. Fuel and electricity consumption associated with geomembrane replacement were also considered assuming consumption rates would be the same as initial installation. For the UASB-Pond site, sludge removal does not have any fuel or electricity requirements since all work is conducted manually. At this site, a valve is manually opened to transfer sludge from the UASB reactor to the drying bed. A summary of inputs equations and the inventory is shown in Table B1 and a summary of the life cycle inventory is shown in Table B2.

Table B1. Summary of model inputs, equations, and inventory items in Bolivia

Model Inputs	Equations	Inventory Items
(1) Existing Condition		
Material Production: Material Type, Properties (kg, m^2, or m^3), Purchase Frequency (qty)	Mass of material (kg) = mass*purchase frequency	Mass (kg), area (m^2) or volume (m^3) of materials (as required)
Material Delivery: Distance (km), Cargo Weight (tons), Mode (vehicle)	Delivery from location (tkm) = mass (tons)*distance (km)	Freight transportation quantity (tkm) of materials delivered
Equipment Operation & Energy Production: Equipment and Fuel Type, Power use (HP), Use Amount (hours), frequency	Electricity from equipment (kWh) = use amount (hrs)*horsepower (HP)*(0.746kW/HP) Fuel use (kg) = use frequency*fuel consumption rate (kg/hr)	Energy used (kWh) and fuel consumed (kg) by on-site equipment
Sludge Disposal: influent TSS (mg/L), Data from equations above	$V_L = 0.00156*Q*SS$ $t_L = 0.25*(V_F/V_L)^a$	Fuel consumed (kg) by on-site equipment
Biogenic Emissions: Influent and effluent BOD_5 (mg/L) or COD (mg/L) data and influent flow rate	CH_4 emission rate ($kgCH_4$/yr) = $1E\text{-}3*Q_{ww}*OD*Eff_{OD}*CF_{CH4}*[(MCF_{ww}*BG_{CH4})(1-\lambda)]^b$	CH_4 (kg) of lagoons & UASB
(2) Energy Recovery Condition		
Natural gas avoided: Percent methane in biogas and Biogenic emission inputs (See above)	Using the density of methane and a 65% average methane content (EPA, 2010) the volume of biogas over the lifespan of the system was calculated. This was then converted to natural gas.	Natural gas avoided (m^3)
(3) Water Reuse Condition		
Pumping power use (HP), amount use (hours), irrigation requirements (m^3/ha/yr), total nitrogen (mg/L), total phosphorus (mg/L)	The citrus water requirements were used to estimate the energy needed to irrigate one hectare of citrus over the lifetime of both treatment systems and compared to energy needed to pump water to irrigate an equivalent amount of citrus with river water.	Fuel consumed (kg) by pumping for irrigation of 1 hectare, total nitrogen (kg), total phosphorus (kg)

[a]V_L=Annual volume of sludge produced (m^3/yr), Q_{mean}=Average flowrate (m^3/day), SS=Influent suspended solids (mg/L) or TSS concentration, t_L=Sludge removal frequency (years), V_F=Volume of the facultative lagoon (m^3). (Oakley, 2006) [b]Where, 10^{-3}=Conversion from (kg/g), Q_{ww}=Wastewater influent flow rate (m^3/year), OD=Oxygen demand of influent as BOD_5 or COD (g/m^3), Eff_{OD}=Removal Efficiency of Oxygen demand, CF_{CO2} or CF_{CH4}=Conversion factor for maximum CO_2 (or CH_4)generation per unit OD (g/gOD), MCF_{ww}= Fraction of influent OD converted anaerobically in wastewater treatment unit, BG_{CH4}=Fraction of carbon as CH_4 in generated biogas (0.65), λ=Biomass yield in wastewater treatment unit. . For anaerobic treatment process, MCF_{ww} = 0.8. For shallow facultative lagoons (<2m deep), MCFww = 0.2. Assume maturation lagoon has same MCFww as facultative lagoon. For anaerobic treatment process, λ=0.1. For shallow facultative lagoon (<2m deep), λ=0. Assume maturation lagoon has same λ as facultative lagoon

An EPA estimation method was used to calculate CO_2 and CH_4 biogenic emissions from the UASB reactor, facultative lagoon, and maturation lagoons (EPA, 2010). Biogenic emissions from facultative and maturation lagoons were calculated using factors for ponds less than 2m deep and emissions from the UASB reactor were calculated using factors for anaerobic treatment of wastewater. Nitrous oxide emissions are considered negligible for these systems due to limited nitrogen removal, whereas biogenic CO_2 is considered negligible (IPCC, 2006). Required inputs to calculate biogenic CH_4 included BOD_5, COD, and flowrate. Average influent flow data (n=4) and average COD and BOD_5 (n=5) entering the facultative lagoon, maturation lagoon or UASB reactor were collected in field from 2007-2011. The measured content of methane (CH_4) ranged from 56-77% (Muga et al., 2009; Verbyla et al., 2013). An assumed methane content of 65% (EPA, 2010) was used to estimate biogenic air emissions and emissions avoided through the recovery of biogas under energy recovery conditions.

The recovery of biogas as a co-product is assumed to eliminate carbon dioxide, while methane emissions from the UASB reactor replace natural gas usage. The amount of natural gas avoided is calculated based on methane production and the energy content of natural gas and methane (Galvin, 2013). This represents the maximum energy offset from produced biogas. Biogas purification infrastructure is not considered in the scope of this study. The 3-Pond system in Bolivia has no recoverable energy.

In Bolivia, citrus water requirements were used to estimate the electricity needed for agricultural irrigation of 350 m^3/ha over the life of the systems. The pumping requirements are based on an irrigation system that transfers water into a 3.78 m^3 (1,000-gallon) tank and subsequently irrigates citrus trees via gravity during the dry seasons only. Average irrigation requirements for citrus were calculated using values provided by a local agricultural engineer and estimates using the Food and Agricultural Organization (FAO) software. CROPWAT 8.0 software uses local data (e.g., temperature, humidity, wind speed, sun hours, evapotranspiration,

Table B2. Life cycle inventory per cubic meter of treated water over 20-year lifespan in Bolivia

Inventory Item	UASB-Pond	3-Pond
Bathroom Construction		
Portland Cement (kg)	1.2E-01	3.4E-02
Ceramic brick (kg)	2.5E-01	9.4E-02
Wood (m^3)	7.1E-05	2.7E-05
PVC (kg)	9.9E-03	4.9E-03
Sanitary ceramics (kg)	1.8E-02	6.6E-03
Transport (tkm)	1.1E-01	3.6E-02
Electricity (kWh)	4.5E-03	1.7E-03
Collection Construction		
Portland Cement (kg)	6.1E-02	4.1E-02
Wood (m^3)	2.8E-05	1.1E-04
PVC (kg)	5.1E-02	1.6E-02
Diesel (kg)	3.5E-02	1.6E-02
Transport (tkm)	4.9E-02	8.6E-02
Electricity (kWh)	3.7E-02	4.1E-03
Pretreatment Construction		
Portland Cement (kg)	1.5E-03	-
Wood (m^3)	7.8E-07	-
HDPE (kg)	1.9E-08	-
Transport (tkm)	2.4E-03	-
Electricity (kWh)	5.8E-05	-
UASB or Facultative Pond Construction		
Portland Cement (kg)	3.5E-02	7.7E-04
Wood (m^3)	1.9E-05	4.5E-07
PVC (kg)	4.2E-04	-
HDPE (kg)	0.0E+00	1.6E-03
Transport (tkm)	2.0E-02	8.4E-03
Electricity (kWh)	2.6E-03	2.0E-03
Diesel (kg)	-	2.2E-02
Sludge Drying Bed Construction		
Portland Cement (kg)	3.3E-03	-
Wood (kg)	2.8E-06	-
HDPE (kg)	8.2E-08	-
Transport (tkm)	7.3E-04	-
Electricity (kWh)	2.7E-04	-

Inventory Item	UASB-Pond	3-Pond
Maturation Pond Construction		
Portland Cement (kg)	6.6E-03	1.6E-03
Wood (m^3)	3.7E-06	3.0E-06
HDPE (kg)	3.2E-03	1.2E-03
Diesel (kg)	2.8E-02	1.7E-02
Transport (tkm)	1.5E-02	6.3E-03
Electricity (kWh)	3.3E-04	1.6E-03
Effluent Structure Construction		
Portland Cement (kg)	2.3E-02	3.0E-04
Wood (m^3)	2.0E-07	1.6E-07
PVC (kg)	2.6E-04	9.1E-05
Transport (tkm)	5.1E-03	1.2E-04
Electricity (kWh)	2.0E-05	1.5E-05
Existing Nutrient Discharge Operation		
Total Nitrogen, TN (kg)	5.18E-02	3.47E-02
Total Phosphorus, TP (kg)	9.40E-03	6.40E-03
UASB or Facultative Pond Operation		
Transport (tkm)	-	5.6E-03
Electricity[b] (kWh)	-	2.7E-04
Diesel[b] (kg)	-	1.3E-03
Electricity[c] (kWh)	-	1.7E-03
Diesel[c] (kg)	-	1.8E-02
HDPE[c] (kg)	-	1.6E-03
CO_2 emissions (kg)	1.2E-01	3.4E-01
CH_4 emissions (kg)	5.0E-02	1.8E-03
Maturation Pond Operation		
CO_2 emissions (kg)	2.4E-01	4.0E-02
CH_4 emissions (kg)	1.3E-02	2.2E-03
Water Reuse Condition Operation		
Electricity[d] (kWh/ha)	6.7E-05	5.4E-05
TN avoided (kg/ha)	3.4E-05	1.8E-05
TP avoided (kg/ha)	6.1E-06	3.3E-06
Energy Recovery Condition Operation		
Natural gas avoided (m^3)	7.1E-02	-
UASB emissions avoided	See above	-

[a] This table excludes items with a contribution less than 1% and select items with a contribution of less than 4% (reinforcing steel, door wood, cast iron). [b] For sludge disposal. [c] For geomembrane replacement. [d] For irrigation pumping.

and rainfall from nearby meteorological stations) to estimate the irrigation requirements for specific crops (FAO, 2012). Pumping electricity needed to meet irrigation requirements are considered, however irrigation infrastructure is not included.

To quantify the benefit of water reuse, agricultural irrigation of reclaimed water was compared to baseline conditions in which river water is used for irrigation. The irrigation pumping energy under baseline conditions is the same as the water reuse condition, however water reuse has an added nutrient benefit, which increase crop yield. Water reclamation has been found to increase crop yield by 10 to 30% (Asano and Levine, 1998; Fatta et al., 2005). This increase in crop yield is assumed to decrease the amount of water needed to irrigate an equivalent amount of crops, thereby decreasing the amount of electricity needed for pumping compared to baseline conditions.

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