SUSTAINABILITY ASSESSMENT OF WASTEWATER TREATMENT PLANTS WITH WATER REUSE FOR URBAN AGRICULTURE: A CASE STUDY IN HYDERABAD, INDIA

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Sustainability Assessment of Wastewater Treatment Plants with Water Reuse for Urban Agriculture in India

Thesis directed by Professor Anu Ramaswami.

ABSTRACT

Wastewater reuse in urban agriculture is a widespread practice in many developing world cites that has many advantages (water savings, nutrient cycling, and livelihoods) and disadvantages (pathogen health risk). Energy use and associated greenhouse gas (GHG) emissions in centralized wastewater treatment plants (WWTPs) can mitigate some of the health risks, however these tradeoffs have not been quantified. The objective of this thesis is to conduct a sustainability assessment of WWTPs with water reuse for urban agriculture in India. Three stages of work were included.

1) The role of water and wastewater (W/WW) infrastructures in urban energy metabolism was explored first. W/WW infrastructures were found to contribute 3 to 16% of community-wide electricity use and greenhouse gas (GHG) emissions for 16 Indian cities; for another 23 the proportion was less than 3%. Energy intensity for drinking water supply and wastewater treatment averaged 1.3±0.7 Wh/gal (n=7) and 0.4±0.2 Wh/gal (n=5), respectively. Energy intensity for water pumping/treatment was more than double that for wastewater, the reverse of cities in Colorado, USA, likely due to poorer source water quality in India.

2) A life cycle assessment (LCA) was conducted of Nallacheruvu (8MGD) WWTP in Hyderabad, India that used upflow anaerobic sludge blanket reactor followed by oxidation ponds, yielding 99% and 81% removal of fecal coliforms and BOD_5, respectively. The LCA showed energy use and GHG emissions of 0.7Wh/gal and 1gCO2e/gal, 48% of the later from on-site electricity use, 41% from methane process emissions, 10% from embodied energy in infrastructure, and 1% from nitrous oxide process emissions. A consequential LCA, conducted using the DAYCENT model for wastewater reuse in urban agriculture, showed only 1% of the nitrogen in treated effluent was reused in urban agriculture, due to land constraints along the flow path of the wastewater. As a result, annual system-wide GHG emissions for untreated and treated wastewaters releases to the riverine system were similar at 2,463mtCO2e and 2,819mtCO2e, respectively. Avoided impacts due to reuse of biogas for electricity and avoided fertilizer each accounts for 5% reductions from the total for treated water.

3) An urban agriculture site study was conducted to assess the impact of pathogen reduction in WWTP on spinach. This was explored in a site study using three different waters: groundwater, treated effluent from WWTP, and untreated water. While *E.coli* in the waters consistently differed by 2-3 orders of magnitude between the three plots, the *E.coli* in the crop measured at the endpoint of the study (harvest conditions) was not significantly different between the groundwater and WWTP plots (t-test $P>0.1$), while the untreated water was slightly higher ($P<0.025$). For *Ascaris*, qualitative results showed little difference between *Ascaris* on crop grown with treated and untreated waters (26-36 eggs/100g spinach), while groundwater-irrigated spinach had lower *Ascaris* levels (9 eggs/100g spinach). Unexpectedly, when the researcher carefully took crop samples, the *E.coli* results compared to farmer-harvested crop were one order of magnitude lower, suggesting recontamination of the crop from farmer handling and contact with soil and contaminated water. The recontamination hypothesis was confirmed ($P<0.1$) by sampling each of the three plots, comparing sanitized handling (n=3) versus farmer handling (n=3) in each plot.

Using data from WWTP LCA and urban agriculture site study, a sustainability assessment showed that treated effluent and untreated surface water were similar in the case of GHG emissions, pathogen risk (15% and 18% probability of disease over one year based on *E.coli* results), yield (20kg/m^2 and 23kg/m^2 for one year) and water saved (0 gallons groundwater used), but varied in terms of economic investment ($97,093 vs $0 per year). The groundwater site had lower GHG emissions, energy use, pathogen risk, but consumed a lot of water (1,125 gallons/m^2/year), yielded only 10% crop at harvest (2kg/m^2) compared to the other plots, and cost approximately $600 per year. If groundwater tables are at risk, then wastewater reuse offers an alternative. While the WWTP technology did not provide as many benefits to urban agriculture as expected, there may be benefits towards cleaning up the Musi River and avoiding groundwater contamination that are beyond the scope of this work. More studies with social actors and institutions are needed to identify sustainability priorities in each community.

This work contributes to:

- The urban metabolism literature by examining energy use and intensity in water/wastewater infrastructures in developing country cities,

- WWTP LCA literature by conducting a first LCA using operating data from an Indian WWTP with water reuse in urban agriculture,

- The urban agriculture literature by completing a first field study of pathogen impacts from treated and untreated wastewater use, and

- The sustainability assessment literature as it links water/wastewater, energy, infrastructure capital investments, urban agriculture, and health.

This abstract accurately represents the content of the candidate's thesis. I recommend its publication.

Signed [redacted]

Anu Ramaswami

DEDICATION

To the strong women who have inspired me.

Also, to my father and mother, who never put limits on what I could do.

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1. Introduction

As city populations grow, their urban metabolism (resource consumption, energy use, and waste generation) also increases (Wolman 1965; Kennedy, Cuddihy et al. 2007). Often in developing nations, cities displace surrounding agricultural land and fresh irrigation water, forcing agriculture downstream of urban riverine/wastewater discharges. This nutrient-rich resource is valuable to farmers who are seeking a widely-available and consistent source of irrigation water for their crops. This practice is not new or rare; in fact, it stems from ancient Greece and today, an estimated 200 million farmers irrigate at least 20 million hectares with raw or partially treated wastewater (Raschid-Sally and Jayakody 2008). This accounts for approximately 8% of total worldwide irrigated land (263 million hectares in 1996), of which two-thirds lies in Asia (Howell 2001). This amount of farmers represents approximately 15% of the total amount of people economically active in agriculture worldwide (FAOSTAT 2009). Some cities where the practice actively takes place today, along with some crops grown and some pathogens associated with wastewater irrigation are listed in Table 1-1 (van der Hoek 2004).

Table 1-1: Examples of cities, crops, and pathogens associated in the practice of wastewater irrigation worldwide

Cities	Crops Grown	Pathogens
Mexico City, Mexico	Spinach	Roundworm (*Ascaris*)
Lima, Peru	Lettuce	Hookworm
Hyderabad, India	Parsley	*E. coli*
Ho Chi Minh, Vietnam	Cilantro	Giardia (*Giardia lamblia*)
Teheran, Iran	Tomatoes	Hepatitis A Virus
Nairobi, Kenya	Potatoes	Typhoid (*Salmonella typhi*)
Kumasi, Ghana	Berries	Cholera (*Vibrio cholerae*)

Because this practice is widespread and legislation is difficult, the question is no longer if wastewater should be used for irrigation, but how it can be made more sustainable and safe (Scott, Faruqui et al. 2004; Van Rooijen, Turral et al. 2005). The next two sections briefly describe the advantages and disadvantages of wastewater reuse for urban agriculture.

1.1 Advantages of Wastewater Reuse for Urban Agriculture

Wastewater reuse for urban agriculture is often considered to provide many benefits. Qualitatively, these advantages are reported to be:

- Conservation of water: Water reused for urban agriculture means that less freshwater is needed, which is important as water scarcity is increasing (van der Hoek, Hassan et al. 2002).

- Nutrient recycling: Wastewater contains nutrients, leading many farmers to prefer wastewater for irrigation because it is thought to increase productivity (Qadir, Wichelns et al. 2007).

Figure 1-1: Urban agriculture in Hyderabad, India

- Avoided fertilizer (Asano 1998): Nutrients in wastewater could save the farmers money and could have the indirect impact of saving energy and greenhouse gases (GHG) (Pitterle and Ramaswami 2009).

- Land treatment of wastewater: Without other treatment options, land application may provide some decrease in surface freshwater contamination (Raschid-Sally and Jayakody 2008).

- Spatial and temporal accessibility of irrigation water: Oftentimes, farmers have better access to wastewater as a source of irrigation water because it is in constant supply in urban and peri-urban areas, even in the dry season. This is because cities are drawing municipal drinking water from outside their boundaries and it is being discharged as wastewater after use (Qadir, Wichelns et al. 2007).

- Decreased need for expensive refrigerated transport or storage facilities: This is most valued in developing countries with hot climates (Qadir, Wichelns et al. 2008).

Figure 1-2: A local farmer heading to market with newly harvested spinach grown with untreated surface water

- Nutrition: Urban agriculture (which is facilitated by wastewater reuse in many developing world cities) provides both farmers and consumers with a local, fresh supply of vegetables (Qadir, Wichelns et al. 2008).

- Better livelihoods: Wastewater is an inexpensive source of water and nutrients allows farming families to grow high-value and high-demand crops like vegetables (Kilelu 2004), which generates more income and raises living standards, therefore allowing for indirect benefits like education (Raschid-Sally and Jayakody 2008).

For these reasons, wastewater is considered a valuable resource for many. The articles/reports above are largely qualitative studies. Many of these benefits, along with savings in energy, greenhouse gas emissions, and water, have not been measured.

1.2 Disadvantages of Wastewater Reuse for Urban Agriculture


Figure 1-3: Flood irrigation with untreated surface water in Hyderabad, India

On the other hand, while there are many advantages, the practice of wastewater reuse in urban agriculture poses public health and environmental problems as water, soil, and crops become increasingly contaminated. Wastewater contains a variety of pollutants such as: salts, metals, metalloids, pathogens, residual drugs, organic compounds, endocrine disruptor compounds, and active residues of personal care products (Qadir, Wichelns et al. 2007). Farmers in developing countries often use water from a polluted stream, diluted wastewater, or untreated sewage directly on crops. Wastewater from any source is seldom treated before being applied to crops

(Qadir, Wichelns et al. 2007). Pollution in wastewater frequently affects soil quality and/or causes acute or chronic diseases. Pathogens, specifically intestinal nematode infections, have been identified as the main threat to human health in the short term (Ensink, Blumenthal et al. 2008).

1.3 Wastewater Treatment Plants (WWTPs) for Sustainable Water Reuse for Urban Agriculture

WWTPs are effective in removing pathogens and other harmful substances from water and have been shown to decrease health risk (Asano 1998). Cities in the developing world are implementing WWTP infrastructure to address this need for treatment of sewage-polluted water. But WWTPs are themselves energy intensive. A joint study by the American Water Works Association Research Foundation (AWWARF) and the California Energy Commission, compiled data on wastewater utilities in the US (Association of Metropolitan Sewerage Agencies, now the National Association of Clean Water Agencies or NACWA and a study in Iowa) and these ranged from 0.8-3.5 Wh/gal in energy use (Carlson and Walburger 2007). Conventional wastewater treatment alone is estimated to consume 3-5% of U.S. electricity (Shizas and Bagley 2004; US EPA 2006). For a typical U.S. municipal energy budget, wastewater treatment is one of the largest at 23% (Means 2003). These energy investments are expected to offer various benefits in terms of pathogen reduction and they may help in more sustainable wastewater reuse for agriculture. Also, overall greenhouse gas (GHG) emissions reductions may be achieved due to WWTP processes and subsequent application of effluent to farmlands.

No literature has been published that explores the fate of GHG emissions when wastewater is reused in agriculture. Some studies have computed GHG benefits of avoided fertilizer when nutrients are reused in agriculture (Pitterle 2008). However, there is high uncertainty associated with direct nitrous oxide (N_2O) emissions from wastewater (IPCC 2006; Del Grosso, Ojima et al.

2009). The DAYCENT model, which has been verified to yield reliable results in N_2O emissions from cropped fields (Del Grosso, Mosier et al. 2005; Jarecki, Parkin et al. 2007), has not been used to estimate emissions from wastewater application to land. Thus, the energy investments in WWTP-related GHG emissions and associated pathogen risk reduction for urban agriculture are unknown.

1.3.1 Sustainability assessment of wastewater treatment plants with water reuse in urban agriculture

Based on the above review, wastewater infrastructure can have multiple and conflicting sustainability impacts: economic benefits to farmer (food production), health benefits to society (pathogen risk reduction in food), GHG emissions (not known whether it will increase or decrease), water reuse (water savings) and monetary cost (increases with more infrastructure). This sustainability quadrant (Figure 1-4) is one way of weighing tradeoffs and has been used by other authors (Pearce and Vanegas 2002; Aubin, Papatryphon et al. 2009). This paper will evaluate these tradeoffs for the three sites used in this study, differing sources of irrigation water: groundwater, treated effluent from a WWTP, and untreated surface water representive of the sewage-contaminated riverine system.

The five corners represent quantifiable environmental and social parameters pertaining to sustainability:

- Irrigation water saved will be measured by the amount of water reused for irrigation and therefore, the equivalent freshwater/groundwater saved.
- Food produced is a parameter that is closely linked with nutrient delivery as well as with farmer livelihoods. It is shown as the inverse as higher impacts are shown as "worse".
- Pathogen risk reduction was determined from lab tests on pathogen content of vegetables.

- Greenhouse gas reduction is based on the full life cycle GHG emissions for delivering the irrigation water to each site.
- Monetary investment is based on infrastructure investments, amortized over their lifetimes.

All plots will be measured against one another for each parameter as a percentage.

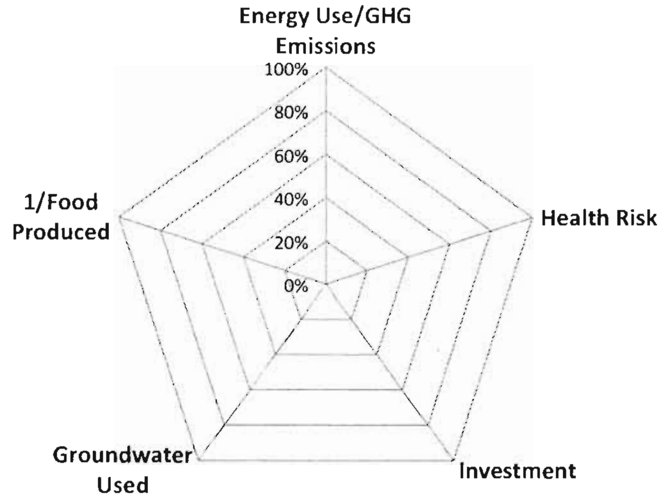


Figure 1-4: Sustainability pentagon for evaluating the tradeoffs of the coupled water-wastewater-urban agriculture system

Although WWTP infrastructure may offer many benefits, its overall role in urban metabolism has not been measured. Therefore, quantification of sustainability tradeoffs to the coupled water-wastewater-urban agriculture system is important at this time when WWTP infrastructure is being built.

1.3.2 Case study selection

Because many location-specific factors affect these tradeoffs, a case study approach was necessary. Hyderabad, India as chosen for the following reasons: centralized WWTP infrastructure is newly implemented (secondary treatment within the last 5 years), wastewater contamination of surface water is ubiquitous, and wastewater-polluted water is reused for urban agriculture. From the case study, data was obtainable on livelihoods (food production), health risks (pathogen content), wastewater reuse (water use/savings), wastewater treatment (energy use and GHG emissions), and wastewater infrastructure (cost).

1.4 Research Goals and Objectives

This study aims to quantify multiple and competing sustainability impacts of implementing WWTP infrastructure in developing world cities with subsequent reuse of wastewater in urban agriculture. The five sustainability metrics are: water use/savings, nutrient supply/food production, pathogen content, energy use/greenhouse gas emissions, and cost of infrastructure.

1.4.1 Phases of Work

This work is implemented in four steps:

1) Role of Water and Wastewater in Urban Energy Metabolism (Ch. 2): a field analysis for India on water supply and wastewater treatment infrastructure-related urban metabolism (energy use and energy-related GHG emissions);

2) Understanding Case Study Area and Infrastructure Components (Ch. 3): description of Hyderabad, India, and its water supply, wastewater treatment, and urban agriculture infrastructures;

3) Wastewater Treatment Plant Life Cycle Assessment (Ch. 4): a WWTP life cycle assessment (LCA) measuring energy and GHG investments in WWTP with and without subsequent water reuse for urban agriculture;

4) Measuring Water Quality and Food Quality Relationships: A Site Study (Ch. 5): a farming urban agriculture study near the Musi River to determine water use, nutrient delivery, and pathogen risk at three sites: one site irrigated with freshwater/ groundwater, one irrigated with effluent from the WWTP, and one irrigated with untreated surface water;

2. The Role of Water and Wastewater Infrastructures in Urban Energy Metabolism

2.1 Introduction

Throughout history and in all parts of the world, it has been well-documented that sewage-contaminated water poses problems to human and environmental health (World Health Organization 2006; Laine, Huovinen et al. 2010). Today, as access to basic sanitation is lacking in many developing countries, practices, such as open defecation and uncontrolled release of sewage (blackwater) to rivers, are polluting riverine systems in major cities of the world (Van Rooijen, Turral et al. 2005; Qadir, Wichelns et al. 2008; Raschid-Sally and Jayakody 2008).

Wastewater treatment plants (WWTP) or sewage treatment plants (STP) are useful in removing pathogens and other pollutants from water. Many rapidly growing cities are therefore installing WWTPs, providing primary, secondary, and sometimes even tertiary treatment, for their communities. WWTPs have an important role in a city's metabolism of water, nutrients, and energy.

2.1.1 Urban Metabolism: Water, Nutrients, and Energy

Urban metabolism studies often focus on a flow across city boundaries of: water, substances (e.g. nutrients, chemicals, food, construction materials, etc.), or energy. Some studies, like Kennedy et al., considers on all of these flows and observes that city metabolism is generally increasing worldwide (Kennedy, Cuddihy et al. 2007).

Some authors have focused on water foot-printing in order to track the inflow and outflow of water in urban regions, incorporating water use for consumption and production of goods and services (Allan 1998; Luck, Jenerette et al. 2001; Jenerette, Wu et al. 2006; Yu, Hubacek et al. 2010; Water Footprint Network 2011). Water footprint methods are being refined to incorporate embodied water in energy flows into cities.

Substance flow analysis can be done for macronutrients like nitrogen (N) and phosphorus (P). A nitrogen balance was done by Lawrence Baker et al. for the central Arizona-Phoenix (USA) ecosystem. They identified natural inputs of fixed N, deliberate human-mediated N inputs (mainly agricultural-related, including fertilizers), and inadvertent human-mediated N inputs (combustion-derived NO_x) (Baker, Hope et al. 2001). Nitrogen was followed through transfers within and among subsystems, and accumulation within the ecosystem. Then, the outputs of nitrogen as deliberate exports (in food and wastewater) and inadvertent exports (NO_x, N_2O, NH_3, N_2, and surface water) were tracked. The overwhelming majority, 88%, of total nitrogen inputs to the ecosystem were found to be human-mediated: deliberate inputs accounted for 52%, while inadvertent inputs made up the additional 36% (Baker, Hope et al. 2001). The largest deliberate effort in removing nitrogen was done by WWTPs, which removed 10% of the input nitrogen.

A few groups are studying urban energy metabolism (Wolman 1965; Huang 1998; Hillman and Ramaswami 2009; Zhang, Zhang et al. 2010). Many sectors consume energy and release GHG emissions, and energy flows are multiplied by emission factor to determine the GHG impacts. Some sectors generate a portion of energy themselves, for example, in the production of biogas from WWTPs. Methods for studying energy flows are also being refined and are including embodied energy and transboundary energy use. Accounting for transboundary items is important to the entire life cycle and scopes are used to categorize in-boundary versus out-of-boundary items. Many groups use scopes to inventory transboundary items (Hillman and Ramaswami 2009; ICLEI 2009; Kennedy, Steinberger et al. 2009).

As discussed previously, wastewater treatment alone is estimated to consume 3-5% of U.S. electricity (Shizas and Bagley 2004; US EPA 2006), but this proportion may be even higher in developing countries where electricity is less common. Due to a lack of energy data, the role of water and WWTP infrastructure in the energy metabolism of developing world cities is not well-known. To achieve a holistic view, urban metabolism studies are needed to quantify the proportion of energy use in urban systems that goes towards water and wastewater infrastructure.

2.2 Data sets used for this study

In order to understand the role of water and wastewater infrastructure on energy use in Indian cities, primary data was obtained from two different sources.

2.2.1 Basic Infrastructure Data

The Indian Ministry of Urban Development (MoUD), Government of India (GoI) completed a study for 2008-9 on service level benchmarking for water supply, wastewater treatment, storm water drainage and waste management. The purpose of the study was to understand what improvements have been made due to GoI's financial assistance towards infrastructure improvements for delivery of municipal services to city residents (Ministry of Urban Development 2010). 28 cities from 14 states participated in this study.

The MoUD study includes the following indicators for water supply: number of connections (both residential and non-residential), volume water produced, source of water supply (% from groundwater and surface water), volume water consumed (both residential and non-residential), water treatment capacity of drinking water treatment plants, volume of treated water storage, distribution pipe length, average pressure, and number of water samples passing the standards tests (Ministry of Urban Development 2010). The water supply data does not include: the type of treatment, the energy used for pumping and treating the water, or the specific sources of water (or distances it travels/is pumped).

For wastewater treatment, the following indicators are included in the MoUD study: the number of properties with access to toilets and the number that are connected to sewers (as only a portion of toilets are connected to the sewer system), the area covered by the sewerage network, the number of WWTPs, the volume of sewage treated, the volume of treated water reused, and the number of effluent samples passing the disposal standards tests (Ministry of Urban Development 2010). The wastewater data in this study lacks: the type of treatment, and the energy used for pumping and treating the wastewater. The report does not specify the proportion of the city's energy or the proportion of the city's GHG emissions that are caused by the water supply and wastewater treatment sectors.

2.2.2 Energy use in water systems and in cities overall

For energy data, a report from ICLEI (Local Governments for Sustainability) was used. ICLEI gathered data on city-wide energy use and GHG emissions within the geographic boundary for 54 South Asian cities, 41 of which are in India (ICLEI- South Asia 2009). This data set used the World Resources Institute's (WRI) scopes 1, 2, and 3 to inventory greenhouse gas emissions. Briefly, scopes 1 and 2 refer to direct GHG emissions and indirect GHG emissions primarily from electricity, respectively, and encompass the traditional accounting method. Scope 3 focuses on indirect GHG emissions and includes critical urban materials that are needed by the city but are produced outside of the boundary.

This data set included scope 1 community-wide energy consumption and related emissions in the residential, commercial, industrial, transportation, waste, and other sectors including the following fuels: liquefied petroleum gas, fuel wood, kerosene, diesel, petrol, light diesel oil, compressed natural gas, auto gas, municipal solid waste, and coal. Also, scope 2 community-wide electricity consumption and related emissions were quantified for residential, commercial, and industrial sectors, as well as for the municipal services of building and facilities, street lighting, water supply and wastewater treatment plant, and others. Data on all of these were not available for every city. More specific data was also provided by ICLEI South Asia and was used to separate energy use for water supply from wastewater treatment.

2.2.3 Intersection of two data sets

To link the water infrastructure with energy infrastructure, data from both reports was needed. Of the 41 cities from the ICLEI report, and the 26 cities from the MoUD report, 11 cities were common. Then, ICLEI had separate water supply for only 6 of these cities and separate wastewater data for only 4 cities. Data from the ICLEI report was obtained for 2007-08 while data from the MUD report was for 2008-09, making them one year different.

Additionally, data from Hyderabad and Delhi was obtained during visits to those two cities. For Hyderabad, data from the Hyderabad Municipal Water Supply and Sewerage Board (HMWSSB) website was used to estimate the energy consumption for pumping of municipal water supply in Hyderabad (HMWSSB 2008). Wastewater data for March 2009- March 2010 was provided by Mr. M. L. Prasanna Kumar at the HMWSSB on one of its WWTPs (Nallacheruvu). This data was scaled up to estimate the total energy consumption of Hyderabad's WWTPs (Kumar 2010). For Delhi, Mr. Ramesh Negi and Mr. Ajay Gupta of the Delhi Jal Board provided data for 2008-09 and 2009-2010 on energy consumption for treatment and distribution (pumping) of the municipal water supply as well as for treatment and pumping of wastewater (Negi and Gupta 2011). In total, un-separated energy data for water infrastructure (water supply and wastewater) was obtained for a total of 13 Indian cities. Separated data will be discussed later.

2.3 Metrics

The three overall metrics used to describe the role of water supply and wastewater infrastructure in urban energy metabolism are found in table 2-1. Data needed to calculate/inform these metrics and the data set compiled are also included.

Table 2-1: Information on metrics and data used for this urban energy metabolism study

Metric			Data Needed to Estimate	Sub-data Needed	Data set
1	Carbon emissions for water infrastructure as a proportion of overall community emissions		Energy-related GHG emissions for water supply and wastewater infrastructure		39 Indian Cities
			Overall energy-related GHG emissions for community/city		
2	Energy use for municipal water supply pumping and treating	per gallon	Energy data on pumping and treating reported separately	Percentage of municipal supply that is groundwater	8 Indian Cities
		per capita	Volume of municipal water provided per capita	How many people are without connections	
3	Energy use for wastewater pumping and treating	per gallon	Energy data on pumping and treating reported separately		6 Indian Cities
		per capita	Volume of wastewater collected per capita	How many people with access to toilets and connected to sewerage system	
		per BOD load removed	Biochemical oxygen demand removal efficiencies		

The methods for these metrics were:

Metric #1. Carbon emissions for energy use in water infrastructure as a proportion of overall community emissions: ICLEI-South Asia (ICLEI-SA) collected data (2007-08) from the engineering and administrative departments of the participating Urban Local Bodies (ULBs) on community-wide scope 1 and 2 energy use (ICLEI- South Asia 2009). Then, ICLEI-SA used this energy data in the Harmonized Emissions Analysis Tool (HEAT) to calculate the equivalent carbon emissions for each sector. ICLEI calculated the percentage of community-wide emissions that were from "corporation" or municipal services provided to the city, of which water infrastructures were a subset (ICLEI- South Asia 2009). Electricity, the primary form of energy used for treatment and pumping of drinking water and wastewater, was the only form used to estimate emissions from water infrastructures.

Metric #2. Energy use for municipal water supply treatment and pumping per gallon and per capita: To determine the energy use per gallon of water, volumes of municipal water provided to city residents was needed and was found in the MoUD report for 2008-09 (Ministry of Urban Development 2010). Electricity data for municipal water supply combing treatment and pumping was provided by ICLEI-SA and 11 of their cities overlapped with data needed in the MoUD report. Electricity data for Delhi was provided by the Delhi Jal Board and was separated for treatment and pumping. For Hyderabad, only water pumping data was available to this researcher (figure 2-5 and table 2-4). Of these 13 Indian cities that data could be gathered for, 8 cities provided separate electricity data for municipal water supply combining treatment and pumping (ICLEI- South Asia 2009). The other relevant sub-data was the proportion of total number of city residences to those that have water tap connections, also given in the MoUD report. The total populations (2008-09) for the ULBs were used to calculate the per capita energy use for municipal water supply (Ministry of Urban Development 2010).

Metric #3. Energy use for wastewater treatment and pumping per gallon, per capita, and per biochemical oxygen demand (BOD) load removed: To determine the energy use per gallon of wastewater, the volume of wastewater collected was obtained from the MoUD report for 2008-09 (Ministry of Urban Development 2010). Electricity data for wastewater combing treatment and pumping was provided by ICLEI-SA and 11 of their cities overlapped with data needed in the MoUD report. Electricity data for Delhi was provided by the Delhi Jal Board and was separated for treatment and pumping. For

Hyderabad, only wastewater treatment data was available to this researcher (figure 2-6 and table 2-5). Of these 13 Indian cities that data could be gathered for, 6 cities provided separate electricity data for municipal water supply combining treatment and pumping (ICLEI- South Asia 2009). To inform energy use per capita for wastewater treatment and pumping, the proportion of properties with connections to the sewer was important (not all toilets are connected to sewers). This data was also found in the MoUD report. The amount of BOD removed could be calculated for both Delhi and Hyderabad as BOD measurements were provided by those two cities.

2.4 Results for Indian cities by three metrics

2.4.1 Carbon emissions for water infrastructure as a proportion of overall in-boundary community emissions

The electricity-related emissions from municipal water supply and wastewater infrastructure as a percentage of the total carbon emissions from 39 communities (or ULBs) are shown in figure 2-1. For the majority of these communities, the proportion of the total electricity-related emissions arising from water infrastructure was less than 6%. However, a few cities had much higher proportional emissions: Lucknow (16%), Shimla (12%), and

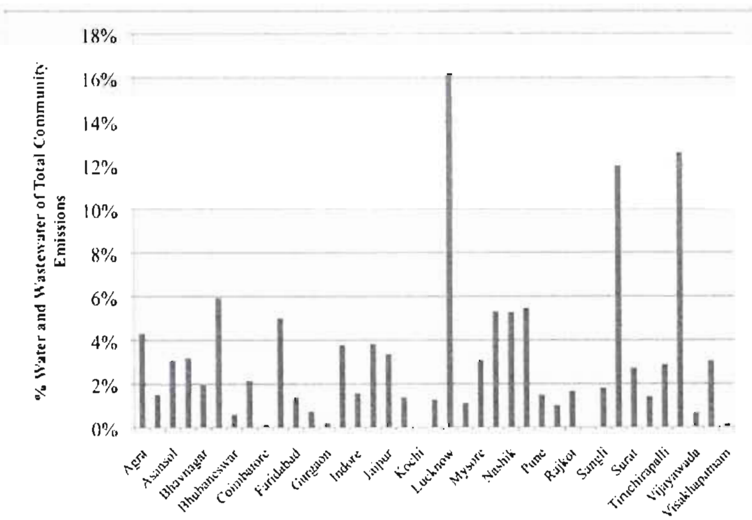
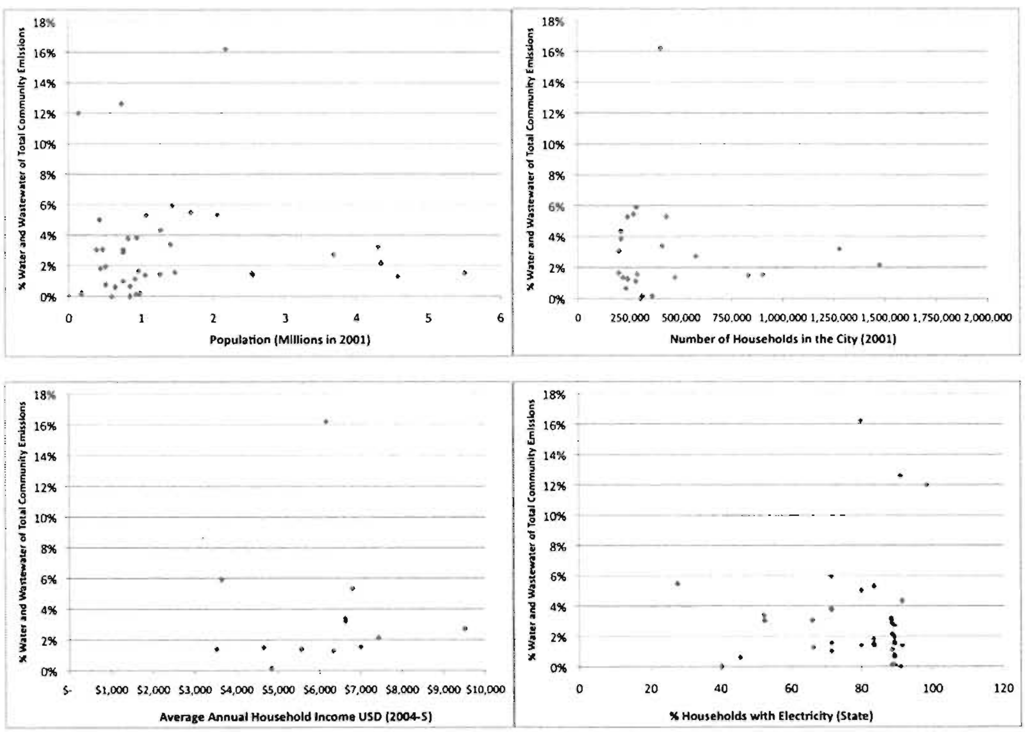


Figure 2-1: Percentage of community-wide GHG emissions associated with in-boundary energy use that are attributed to water and wastewater infrastructure

Trivandrum (13%) (ICLEI- South Asia 2009). Trends that may explain these results were explored in figure 2-2 A-E.

The percentage of emissions from water and wastewater infrastructure were plotted against: city population in 2001 (figure 2-2A) (ICLEI- South Asia 2009), number of households in the city (figure 2-2B) (Ministry of Home Affairs 2001), average household income (figure 2-2C) (TRENDSnIFF 2008), percentage of households with electricity (state percentages) (figure 2-2D) (Ministry of Health and Family Welfare 2007), human development index



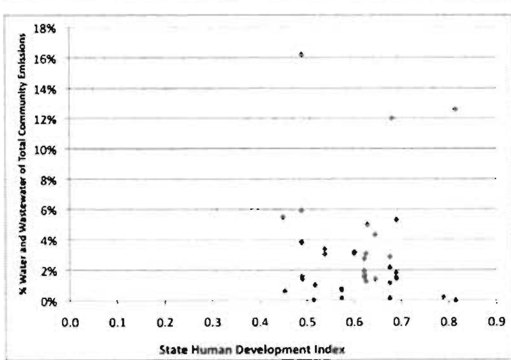


Figure 2-2: Percentage of emissions from water and wastewater infrastructure versus A) population of Indian cities; B) number of households in Indian cities; C) average household income in Indian cities ($USD); D) percentage of households with electricity (Indian state average); or E) human development index (Indian state average).

(state data from 1981) (figure 2-2E) (Government of Meghalaya Shillong 2008).

No significant trends emerged. More reliable household-level data may offer more answers, but currently, the proportion of energy-related GHG emissions for water supply and wastewater infrastructure in cities can not be well-explained by the proxies used here to represent urbanization.

Although these carbon emissions are large, they did not include WWTP process emissions. As described later, process emissions can also be large. Therefore, emissions from WWTP could be doubled.

2.4.2 Electricity use in water and wastewater infrastructure

Electricity use and energy-related GHG emissions for the combined water and wastewater infrastructures can be affected by: the amount of water pumped from various sources, coverage of water supply and wastewater infrastructure, percent of homes served by taps and sewers, and the amount of wastewater collected for treatment. Figure 2-3 describes the water and wastewater infrastructure for Hyderabad, India, while tables 2-2 and 2-3 describe the 13 cities for which the same infrastructure data was available (population and city area for these cities are shown in table 2-4). The Ministry of Urban Development report provides more information on data quality ratings and

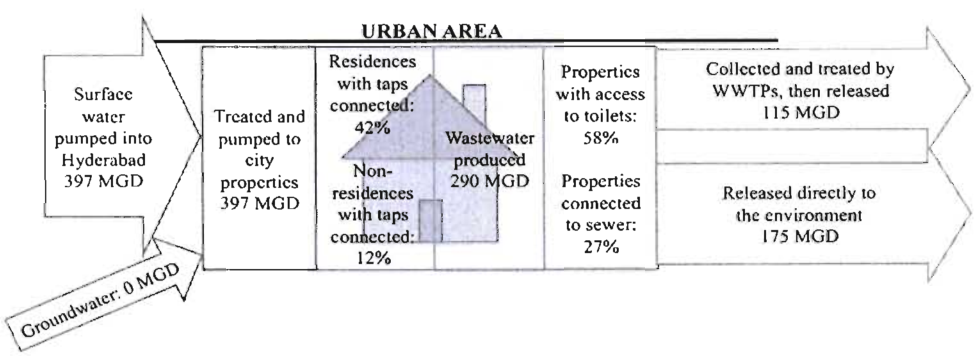


Figure 2-3: Water and wastewater flows through Hyderabad

more information on water and wastewater infrastructure.

Figure 2-3 shows that far more water is supplied to a city than is treated in WWTPs. This explains why, on an aggregate basis, the majority of energy is used for water supply treatment and pumping.

Table 2-2: Municipal water supply profile for 6 Indian cities (Source: Ministry of Urban Development 2010).

Community 2008-09	Surface Water	Ground-water	Residential Water Connections	Non-residential Water Connections	Volume Water Produced	Water Supply Consumed
	% of Municipal Water Supply		% of Residences	% of Non-residences	Million US Gallons/ day	
Ahmedabad	88%	12%	45%	9%	244	171
Delhi	88%	12%	47%	16%	971	809
Guntur	100%	0%	48%	15%	20	9
Hyderabad	100%	0%	42%	12%	397	289
Shimla	97%	3%	32%	117%	9	7
Tiruchirapalli	100%	0%	45%	11%	24	16

Table 2-3: Wastewater profile for 6 Indian cities (Source: Ministry of Urban Development 2010).

Community 2008-09	Properties with Access to Toilets	Properties Connected to Sewer	Sewerage Network Coverage	Volume Wastewater Produced	Volume Wastewater Collected/ Treated	Wastewater Collected/ Treated
	% of Total Properties		% of Community Area	Million US Gallons/ day		% of Wastewater Produced
Ahmedabad	78%	60%	74%	137	89	65%
Delhi	59%	41%	48%	743	467	63%
Guntur	72%	12%	25%	7	nil	nil
Hyderabad	58%	27%	48%	290	115	40%
Shimla	97%	74%	79%	5	1	16%
Tiruchirapalli	91%	23%	25%	22	15	67%

The water flows and infrastructure profiles (figure 2-3 and tables 2-2 and 2-3) can help to explain the differences in energy consumption for water supply and wastewater in cities. Figure 2-4 displays cities with separated data and compares the total electricity used for water supply to that used for wastewater (Note: Data for Hyderabad includes energy use for water supply pumping and wastewater treatment only, while data for Guntur and Tiruchirapalli includes energy use for wastewater pumping only).

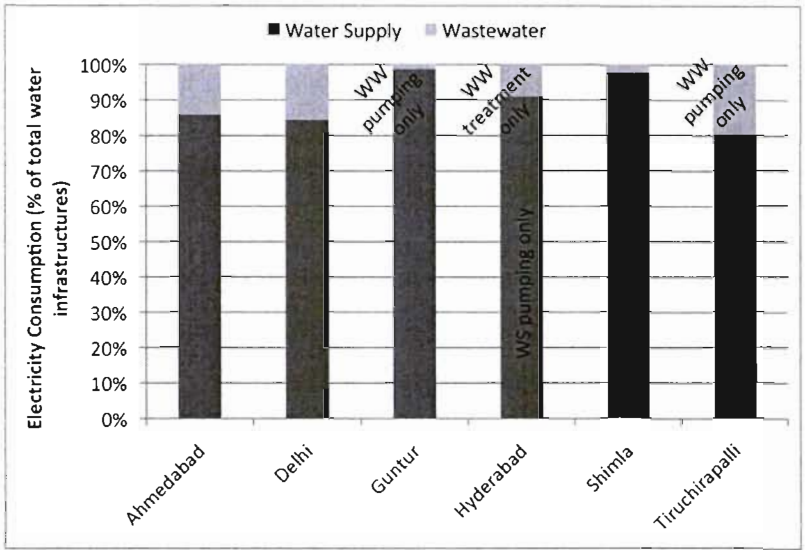


Figure 2-4: Percentage of electricity used for water supply (WS) infrastructure in comparison to wastewater (WW) infrastructure.

In the case of the two cities using the lowest relative amount of energy for wastewater treatment, Guntur and Shimla, this result is expected per the infrastructure profiles. For Guntur, their WWTP is commissioned but not yet running (as of 2008-9), and therefore, no wastewater was being collected or treated. For Shimla, only 16% of the total produced wastewater was collected and treated, which is surprising due to their higher amount of properties connected to sewers and larger percentage of community area covered by the sewerage network coverage.

2.4.3 Energy use for municipal water supply treatment and pumping per gallon and per capita

This section focuses on the energy used for both treatment and pumping of municipal water supply that is distributed to the city residents. Often, data was not available for both treatment and pumping, or it was aggregated together as a total. Figure 2-5 represents the electricity consumption per gallon of municipal water supply. When the outlier (Shimla) was removed from Figure

2-5, average total energy use for water supply treatment and pumping was 1.26±0.68 Wh/gal (n=7).

The cities for which there was not separate data available for water supply treatment and pumping are shown in blue. The two cities for which there was separate data were Delhi and Shimla. Because Shimla is a hill station in the Himalayan Mountains, there is a high amount of energy consumed to pump water uphill to homes. Also, water is sourced from long distances (Ranjan Kumar Guru 2011). In the case of Delhi, treatment energy dominates the total electricity use per gallon for municipal water supply. This could be explained by the large population living upstream from Delhi that is suspected to be the contributing to the pollution in the surface streams. Unfortunately, the types of treatment at the municipal water plants are not included in the ICLEI or MoUD reports.

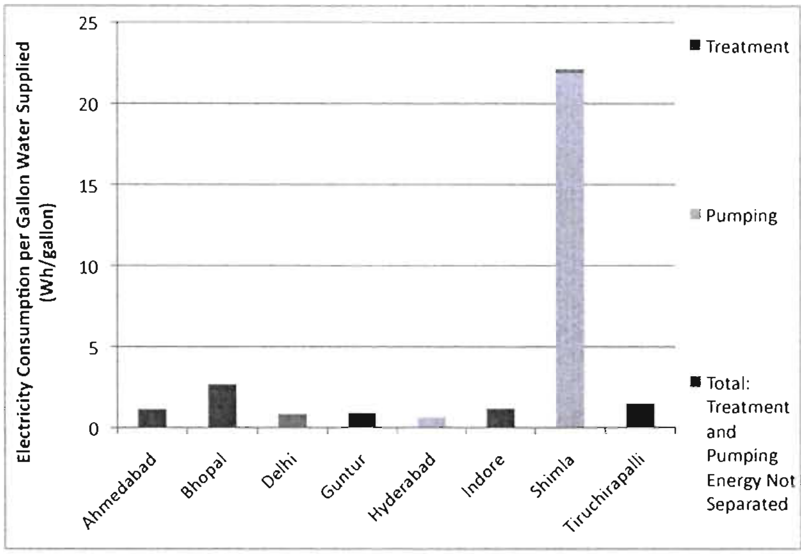


Figure 2-5: Energy use per gallon of municipal water supply (treatment and pumping separated where available)

Again, Shimla stands out with the highest energy user per capita for water supply in table 2-4. Because Shimla is in a mountainous area and water must be pumped up steep gradients, the energy use for supply per person is expected to be high.

Table 2-4: Energy use for municipal water supply for 13 Indian cities.

			Municipal Water Supply								
				Electricity Use					Electricity per Gallon		
	Popu-lation Million 2008-09	City Area	Ground-water in Muni-cipal Water Supply	For Treat-ment	For Pump-ing	Total	Per Capita	Volume Water Prod-uced	Treat-ed	Pump-ed	Total
Community		km^2	%	Million kWh			**kWh/ capita**	Million US Gallons / day	**Wh/gallon**		
Ahmedabad	5.6	466	12%	102		102	**63**	244			**1.14**
Bengaluru	7.8	793	0%					246			
Bhopal	1.8	284	5%	78		78	**42**	79			**2.68**
Bhubaneswar	1.1	135	16%					71			
Delhi	17.8	1,397	12%	274	23	297	**17**	971	0.77	0.06	**0.84**
Guntur	0.6	63	0%	6		6	**11**	20			**0.90**
Hyderabad	7.6	617	0%		91	91	**12**	397		0.63	**0.63**
Indore	2.0	214	12%	21		21	**11**	49			**1.18**
Nashik	1.6	259	0%					91			
Raipur	1.0	154	15%					39			
Shimla	0.2	20	3%	0.8	74	75	**393**	9	0.23	21.9	**22.1**
Tiruchirapalli	0.8	147	0%	13		13	**16**	24			**1.48**
Trivandrum	1.0	142	0%					59			

2.4.4 Energy use for wastewater treatment and pumping per gallon, per capita, and per BOD load removed

This section focuses on the energy use for both treatment and pumping of wastewater. For some communities, data was not available for both treatment and pumping, or it was aggregated together as a total. Figure 2-6 and Table 2-5 represent the electricity data collected. When the outlier (Shimla) was removed from Figure 2-6, average total energy use for wastewater treatment and pumping was 0.41±0.18 Wh/gal (n=4).

The cities for which there was not separate data available are shown in blue. The two cities from which there was separate data are Ahmedabad and Delhi. Guntur and Tiruchirapalli only had wastewater pumping data, while Hyderabad only had wastewater treatment data. Unfortunately, the types of treatment at the wastewater treatment plants are not included in the ICLEI or MoUD reports.

For Delhi, the energy use for wastewater is highest overall, but not per capita or per gallon (Table 2-5). Per capita, Ahmedabad has the highest energy use

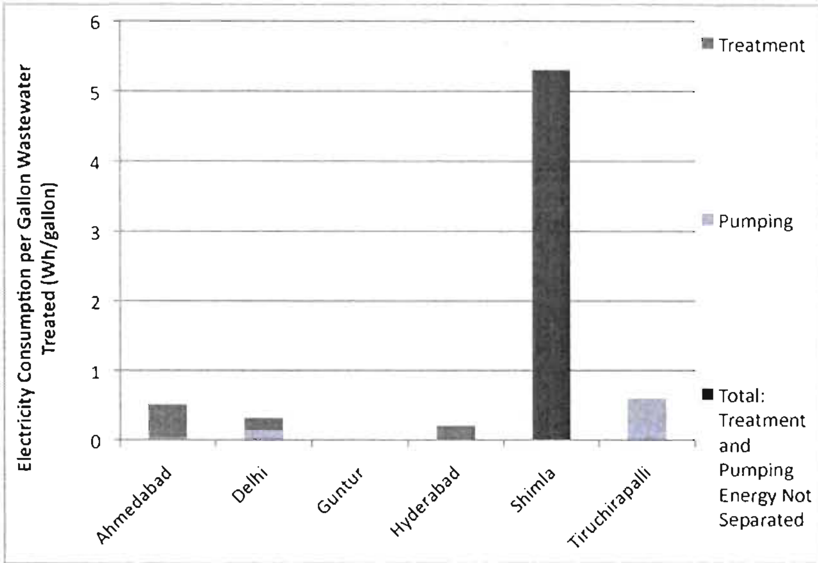


Figure 2-6: Energy use per gallon of wastewater (treatment and pumping separated where available)

for wastewater, largely due to a high amount used in treatment. This is to be expected because Ahmedabad collects 65% of its produced wastewater (one of the highest in table 2-3). Per gallon wastewater treated, Shimla has the highest overall, and may be explained by a high energy requirement for pumping up hills, as with drinking water. Because the data is not separated for Shimla, this cannot be further described at this time.

Last, energy use per milligram of BOD (biochemical oxygen demand) was calculated for Delhi and Hyderabad. These energies were 0.7 and 0.4 Watt-hours/gram (Wh/g) BOD removed, respectively (table 2-5). To benchmark these studies, WWTPs in the US used between 1.5 to 9.8 Wh/g BOD removed using various technologies such as trickling filters, attached growth, activated sludge, and advanced wastewater treatment processes with and without nitrification (Pitterle 2008). Therefore, energy use for removal of BOD was much lower in Indian WWTPs when compared to US WWTPs.

Table 2-5: Energy use for wastewater treatment and pumping for 13 Indian cities.

	Wastewater									
	Electricity Use					Electricity Use per Gallon				
	For Treat-ment	For Pump-ing	Total	Per Capita	Volume Treated	Treated	Pumped	Total	BOD load removal effi-ciency	Energy use per BOD load removal
Community	Million kWh			**kWh/ capita**	Million US Gallons / day	**Wh/gallon**			mg/ gallon	**kWh/ mg BOD removed**
Ahmedabad	16	1	17	**10**	89	0.48	0.03	**0.51**		
Bengaluru					97					
Bhopal					7					
Bhubaneswar					1					
Delhi	32	23	55	**3**	467	0.19	0.13	**0.32**	438	**0.73**
Guntur		0.07	0.07	**0.11**	0					
Hyderabad	9		9	**1**	115	0.21		**0.21**	522	**0.40**
Indore					22					
Nashik					37					
Raipur					0					
Shimla	2		2	**9**	1			**5.31**		
Tiruchirapalli		3	3	**4**	15		0.60	**0.60**		
Trivandrum					0					

Total electricity use and per capita electricity use for both water and wastewater infrastructures is shown in table 2-6.

Table 2-6: Energy use for total water infrastructure (water supply+wastewater) for 13 Indian cities.

	Total Water Infrastructure	
Community	Total Electricity Use Million kWh	Per Capita Electricity Use kWh/ capita
Ahmedabad	123	22
Bengaluru	320	41
Bhopal	78	42
Bhubaneswar	10	10
Delhi	351	20
Guntur	7	11
Hyderabad	100	13
Indore	21	11
Nashik	42	26
Raipur	22	22
Shimla	76	402
Tiruchirapalli	16	20
Trivandrum	46	48

2.5 Comparison between India and US

A wide variation in energy use for drinking water supply and wastewater utilities is seen in both India and the US. An AWWARF study compiled data from other studies (AWWA Water:/Stats database and studies done in Iowa and Wisconsin) and found that total energy use for drinking water utilities ranges from 0.3-3.8Wh/gallon (Carlson and Walburger 2007). As mentioned in Chapter 1, this group also compiled data on wastewater utilities in the US and found that energy use for wastewater utilities ranged from 0.8 to 3.5 Wh per US gallon wastewater treated (Carlson and Walburger 2007). However, when energy use in water supply and wastewater treatment is compared city to city, energy use for drinking water utilities is usually about half of that for wastewater utilities. A research team at UCD has compiled a fairly good data set on drinking water and wastewater treatment for cities in Colorado. Durango and Westminister are shown in Table 2-7.

Table 2-7: Total energy use in water and wastewater utilities in the US and India

	Utility / City	Water Supply	Wastewater
		Wh/gallon	
US	Durango, CO	0.9	3.4
US	Westminister, CO	0.6	1.0
India	Ahmedabad	1.1	0.5
India	Delhi	0.8	0.3

A lower proportional energy use per gallon for drinking water treatment in the Colorado cities is reflective of the source water quality. In India, it is thought that more pumping is needed for chlorination at the drinking water plants to disinfect a higher concentration of pollution in the water. Also, in India, there has been a preference for wastewater treatment technologies that require low energy, like the upflow anaerobic sludge blanket (UASB) (Tare and Nema n.d.). In the US, a higher energy use in wastewater treatment could be attributed to more energy consuming processes and higher levels of treatment, when compared to India.

When total community emissions from a city in the US (Broomfield, CO) is compared to a city in India (Bhopal), the larger proportion of energy use for water and wastewater is evident for India.

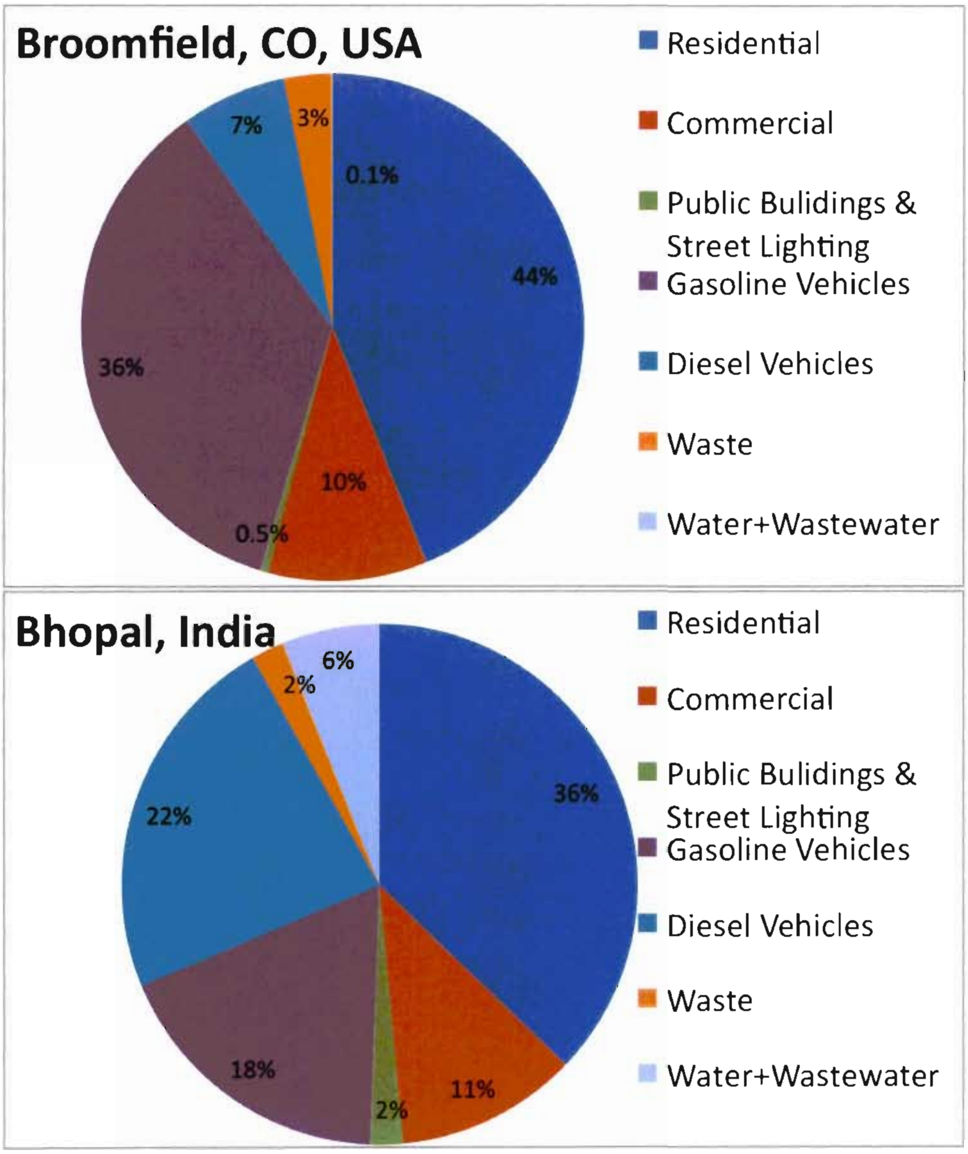


Figure 2-7: Community-wide emissions for Broomfield, Colorado, USA and Bhopal, India.

While there are many differences between these two cities, one notable difference is that water and wastewater contributes a larger proportion of community-wide emissions for Bhopal, India when compared to Broomfield, Colorado, USA.

2.6 Insights and Recommendations for Future Work

From this chapter, we can conclude that energy use and related GHG emissions are a large proportion of the community-wide total. For most communities, more than 1% of the total energy-related emissions were from water and wastewater infrastructures, and this proportion was as high as 16%. With process emissions, this sector could double these emission to contribute 2-32% to community-wide GHGs. On a per capita basis, more energy is invested in drinking water treatment than wastewater treatment overall in India cities. In the next chapters, a case study approach will be used in which a life cycle assessment of a WWTP with linkages to urban agriculture will be discussed.

Future recommendations that would enhance this study are to gather more information on: distances over which water is supplied (both horizontal and vertical), types of treatment used, source water quality, and treated effluent wastewater quality.

3. Understanding Case Study Area and Infrastructure Components

This chapter describes the setting of the study in Hyderabad, India, its riverine system, its WWTPs, and its areas of urban agriculture.

3.1 City Description of Hyderabad, India

3.1.1 Population and Demographics

Hyderabad, located in southern India, is the capital city of the state of Andhra Pradesh. The twin cities of Hyderabad-Secunderabad (herein referred to as Hyderabad) sit roughly in the middle of the country at 526 meters above sea level. Hyderabad is ranked as the 6th largest city in India (World Gazetteer 2010) and the 36th largest city in the world (City Mayors Statistics 2009). The population in 2010 (calculated based on the 2001 census) was 4.1 million (World Gazetteer 2010), while greater Hyderabad was estimated at 7.6 million people (Jacobi 2009; Ministry of Urban Development 2010). The greater Hyderabad area is expected to house 10.5 million residents by 2015 (Sustainable Hyderabad 2006). The urban population of the state is 27.08%, similar to that of India as a whole at 27.78% (Centre for Good Governance 2008). Consequently, 70% of the population is scattered throughout rural areas of the state and make their living as farmers (Mercy Corps 2004). The per capita

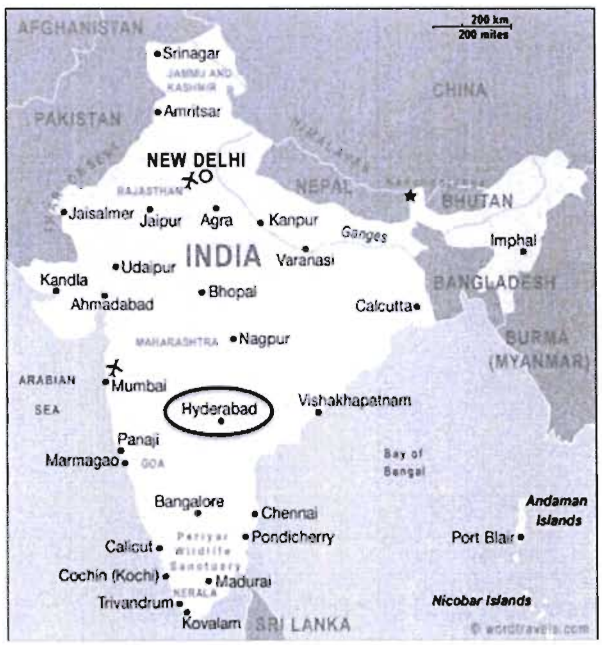


Figure 3-1: Map of India

income is approximately $1.75/day and rising (+13.04% in 2006-2007) while farmer income has seen negative growth (-6.56% in 2006-2007) (The Hindu 2007).

3.1.2 Climate

Hyderabad sits on the Deccan Plateau in the center of India. From May 1, 2009-April 30, 2010, Hyderabad received an average yearly rainfall of 1.6 mm/day (range 0-101.1 mm/day) (Rao 2010). Over the same time period, Hyderabad's average minimum daily temperature was 20.7°C (range 10.0-28.7°C) and average maximum daily temperature was 34.1°C (range 26.0-43.4°C) (Rao 2010). The monsoon season is approximately June-August.

3.2 Water Supply in Hyderabad

3.2.1 Musi River

Hyderabad sits close to the border of the Krishna and Godavari River basins. The Musi River, which flows through the urban center of Hyderabad, is a minor tributary of the Krishna River. West of the city are two dams, Himayat Sagar and Osman Sagar, which were constructed in the early 1950s to regulate the upper catchment of the Musi and provides the 1.1 million residents with 3.5 million cubic meters (MCM) per month of water (Van Rooijen, Turral et al. 2005). Virtually all of this supplied water is consumed within Hyderabad, so the river has little natural flow downstream of the city, but contains a large amount of wastewater. About 150 km downstream of Hyderabad, the Musi River sub-basin drains into the lower potion of the Krishna River (Van Rooijen, Turral et al. 2005). Within the urban area of Hyderabad, the Musi has poor water quality: it "contains raw sewage, chemicals, oils, and other trash", it has "no plant or animal life", it has "a bad odor", and "contact with it is harmful to human health" (Devi, Samad et al. 2009). Hyderabad Municipal Water Supply and Sewerage Board (HMWSSB) authorities measure Musi water quality parameters, and data from 33 tests was provided for the time period between July 2009- March 2010. In table 3-1, this data is compared with the water quality criteria from the Central Pollution Control Board (Ministry of Environment and Forests, Government of India) for the river to be a drinking water source after treatment and disinfection (Central Pollution Control Board 2008). The Musi is not meeting these standards and is not even suitable for boating (Devi, Samad et al. 2009).

Table 3-1: Average Musi River quality (2009-10) compared with India Central Pollution Control Board water quality criteria for drinking water sources before treatment

Parameter/ Water Source	pH	DO [mg/L]	BOD_5 [mg/L]	COD [mg/L]	TSS [mg/L]	VSS [mg/L]	Coliforms* [MPN/ 100mL]
	±standard deviation (n=# tests)						
Musi River	7.3± 0.5 (n=33)	2.1± 2.2 (n=33)	155± 71 (n=33)	402± 176 (n=33)	163± 40 (n=16)	53±22 (n=16)	FC: 418,176 ±139,915 (n=16)
Drinking water source used in conventional treatment and disinfection	6-9	≥ 4	≤ 3	N/A	N/A	N/A	TC: ≤ 5,000

*FC= Fecal Coliforms; TC= Total Coliforms; MPN= most probable number

Downstream of Hyderabad, the water quality of the Musi River improves significantly due to natural processes that occur during settling, aeration, microbial activity, etc., over long distances (Van Rooijen, Turral et al. 2005; Ensink, Blumenthal et al. 2008).

3.2.2 Cross Watershed Transfers

In the 1960s, water from outside of the local catchment area was diverted from the Godavari River Basin, specifically from its tributary, the Manjira River (Celio, Scott et al. 2010). The Singur Dam was built in 1991 to regulate Manjira River water for Hyderabad and it doubled the volume of available water to 18 MCM per month (Van Rooijen, Turral et al. 2005). In 2003, Hyderabad began receiving Krishna River water pumped over 135 km and

400 m in elevation from the Nagarjuna Sagar reservoir, which provided 10 MCM per month (Van Rooijen, Turral et al. 2005; Celio, Scott et al. 2010). Future plans by the HMWSSB, who govern water supply and sewerage for Hyderabad, include further extracts from the Krishna and Godavari Rivers.

3.2.3 Groundwater Withdrawals

Currently, private groundwater withdrawals is estimated to be about 10% of the urban water supply (about 3.3 MCM per month) (Van Rooijen, Turral et al. 2005). However, the municipal water supplied by the HMWSSB does not include groundwater (Ministry of Urban Development 2010). Rapidly growing competition from agriculture and the urban-industrial sector will continue to put stress on the already scarce surface and groundwater resources for Hyderabad and surrounding regions (Van Rooijen, Turral et al. 2009; Celio, Scott et al. 2010; Venot, Bharati et al. 2010; Venot, Reddy et al. 2010).

3.3 Wastewater Infrastructure

For Hyderabad, 80% of the water supply used by people is released as sewage (Ramachandraiah and Vedakumar 2007). According to a Ministry of Urban Development report, 40% of the produced wastewater in Hyderabad is being collected and treated before discharged into the Musi (Ministry of Urban Development 2010). This leaves an average of 175 million gallons of untreated wastewater entering the riverine system every day. For most of the year (in the dry season), the Musi River would not flow without the input of sewage water (Van Rooijen, Turral et al. 2005; Ramachandraiah and Vedakumar 2007).

3.3.1 Wastewater Treatment Plant Technology

Within the last 5 years, Hyderabad has been implementing sewage treatment plants in efforts to treat all of the water entering the Musi to secondary level (Devi, Samad et al. 2009). India is unique in that it has favored UASB technology more than any other country in the world (Khalil, Sinha et al.

2008). UASB reactors were selected in India for the following unique characteristics when compared to other WWTP technology: low capital costs, low energy requirements, low O&M costs, lower skills required for O&M, lower sludge production, and potential for energy recovery and biosolids generation (Khalil, Sinha et al. 2008; Tare and Nema n.d.). However, these characteristics were originally determined for treatment of high strength industrial effluents and may not be as attractive for domestic sewage treatment, as UASB reactors only partially treat the wastewater and may make more problems for the next steps in the WWTP (Tare and Nema n.d.). As described by Heffernan et al, shortcomings in design, construction, and operator maintenance greatly contribute to inferior UASB performance in treating sewage (Heffernan, Lier et al. 2011). Additionally, discharges of industrial effluents into sewage drains flowing into the WWTPs offer many challenges due to the toxic materials and sulphate (contributing to immediate oxygen demand) content of the wastewater (Tare and Nema n.d.).

Among other issues described by Tare and Nema, two notable characteristics of UASB effluent are: BOD will not be lower than 70-100mg/L due to limitations of the UASB reactor, and UASB effluent is highly anoxic and exerts a high immediate oxygen demand on the receiving water body or land. A second stage of aerobic treatment, like that found at Nallacheruvu (the case study WWTP), can lower the BOD and COD by 50% and can increase the dissolved oxygen by 50%, but costs in infrastructure and operations are increased (Walia, Kumar et al. 2011; Tare and Nema n.d.).

Three completed WWTPs that use UASB reactors are located in the south-east area of Hyderabad (Amberpet, Nagole, and Nallacheruvu, while Attapur is not yet complete and is south-central) (see table 3-2). Nallacheruvu WWTP is the case study site for this study.

Table 3-2: Capacity and actual treatment volumes of WWTPs in Hyderabad, India

WWTP	Annual Average Capacity MGD	Annual Average Actual MGD	Ratio
Amberpet	90	66*	0.57
Attapur	13	10*	0.09
Nagole	45	34*	0.29
Nallacheruvu	8	5	0.05
TOTAL	156	115	

* Actual MGD for Ambertpet, Attapur, and Nagole was estimated based on ratio of each capacity to the totals and data from Nallacheruvu WWTP

The ratios in table 3-2, determined from data provided by NWWTP authorities, were used in Chapter 4 to scale up data to the city level when totals for Hyderabad were not known.

The building of the Nallacheruvu WWTP in 2007 displaced urban farmers that had been farming in the area for up to 40 years (McCartney, Scott et al.

Figure 3-2: Urban agriculture adjacent to Nallacheruvu WWTP

2008). Because these farmers used surface water to irrigate their crops, this area has a long history of wastewater contamination in both soil and groundwater. Today, adjacent to the Nallacheruvu WWTP, farmers grow crops such as spinach, coriander, mint, chilies, papaya, amaranth, fenugreek, fennel, and others.

3.4 Urban Agriculture

Downstream of Hyderabad, the Musi River is used extensively for irrigation, with nearly 40,000 hectares of farmland being irrigated from the river (Hamilton, Stagnitti et al. 2007). This has resulted in severe groundwater pollution and an overall long-term decline in the productivity of wastewater-irrigated lands by more than 50 percent (Devi, Samad et al. 2009). A few researchers have studied wastewater reuse in Hyderabad and the effect on the environment and the people (Gopal 2004; Sustainable Hyderabad 2006; Srinivasan and Reddy 2009). The International Water Management Institute has pioneered much of this work in Hyderabad and throughout the world (Buechler and Devi 2002; Devi, Samad et al. 2009; Jacobi 2009). The Resource Centres on Urban Agriculture and Food Security (RUAF) are also active in Hyderabad and globally, with the primary aim to promote and institutionalize urban agriculture processes in cities (RUAF 2010).

3.4.1 Soil Characteristics

Indian soils are generally grouped into two types, referred to as red and black. Red soils, or alfisols (lixisols by FAO classification), are mineral soils with low silt to clay ratio due to a history of strong weathering in wet tropical and subtropical regions (Blokhuis, Bouma et al. 1991; Bhattacharyya, Chandran et al. 2007). Today, they predominately occur in monsoonal and semi-arid regions.

The soils present in this site study were black soils, or vertisols. These mineral soils were conditioned by their parent material, expanding clay, and most occur in semi-arid tropics. They are known to be finely textured and have poor internal drainage, which makes them productive only when managed (Blokhuis, Bouma et al. 1991).

3.5 Discussion

Hyderabad was an ideal location for this study because this researcher had access to urban agriculture, researchers and laboratory facilities at the International Water Management Institute (IWMI) and the International Crop Research Institute for the Semi-Arid Tropics (ICRISAT), and newly implemented WWTPs, making a study based on WWTPs and urban agriculture possible. Next, the case study at a WWTP in Hyderabad will be discussed.

Figure 3-3: Musi River in Hyderabad, India near the outfall of Amberpet WWTP

4. Wastewater Treatment Plant Life Cycle Assessment: Nallacheruvu WWTP in Hyderabad, India

4.1 Introduction

Many developing cities are currently installing centralized wastewater treatment plant (WWTP) infrastructure. As discussed in chapter 2, WWTP processes can be resource intensive in terms of energy use and energy-related greenhouse gases (GHGs), as well as methane (CH_4) and nitrous oxide (N_2O). Direct emissions of N_2O and CH_4, both potent greenhouse gases, can vary by the processes used within the WWTP and subsequent emissions can vary by whether the water is reused in agriculture or released to rivers. A life cycle assessment (LCA) is needed to quantify the water quality improvements achieved with various levels WWTP infrastructure investments that consume energy and release GHGs.

Many life cycle assessments of WWTPs have been done in developed countries, e.g. Australia (Foley, Haas et al. 2005), Canada (Sahely, MacLean et al. 2006), France and Switzerland (Houillon and Jolliet 2005), Germany (Remy and Ruhland 2006), Portugal (Machado, Urbano et al. 2006), Spain (Vidal, Poch et al. 2002; Hospido, Moreira et al. 2004; Munoz, Peral et al. 2007), Sweden (Palme, Lundin et al. 2005), UK (Dixon, Simon et al. 2003), and USA (Murray, Horvath et al. 2008; Pitterle 2008). The focus of these studies range from process specific to whole plant energy and GHG LCA. In general for these developed world LCAs, embodied energy and related GHG emissions are low in comparison to the end-use energy and related GHG emissions. The only LCA study from the above group to use actual WWTP operating data to quantify end-use, embodied, and avoided energy impacts was done by Pittlerle (2008). From Pitterle's work, annual end-use energy for a WWTP in Denver, CO, USA, was 2.3Wh/gal (1.3gCO_2e/gal), while embodied energy was 1.7Wh/gal (0.4gCO_2e/gal). A WWTP benchmarking study in the US also reported end-use electricity at 1 to 5 Wh/gal (Pitterle 2008).

In contrast, for the developing world, there have been very few LCAs of WWTPs. Murray et al. published a hybrid-LCA of sewage sludge treatment in China, but did not include processes to treat wastewater before or after sludge removal (Murray, Horvath et al. 2008). A study for a South-Asian WWTP included energy use for construction activities, process operations, and materials production, but did not quantify GHG emissions (Khan, Aijun et al. 2008). For India, the only study on WWTP energy consumption used model data to determine energy use for a WWTP serving 2,670 residents of a rural Indian village, who generate 39,626 total US gallons wastewater per day. They found that the WWTP would use 62.5kWh/day, or approximately 8.5 kWh/person/year or 0.6 Wh/gallon treated (Devi, Dahiya et al. 2007). When compared to the wastewater infrastructure electricity use in chapter 2, which ranges from 3 to 10 kWh/person/year, or 0.2 to 5 Wh/gal, the result from Devi et al. is realistic. As WWTPs are being implemented in countries like India, Brazil, and Colombia, UASB technology is being recommended because they cost less and use less energy (Khalil, Sinha et al. 2008; Bdour, Hamdi et al. 2009). Consequentially, other components, such as N_2O and CH_4 process emissions, can become very important in life cycle impacts for developing countries.

N_2O emissions from wastewater and WWTP are of interest because of their high global warming potential (298mtCO_2e/mtN_2O) and the high uncertainty in their release. N_2O is emitted naturally from wastewater due to microbial-facilitated nitrification and denitrification. WWTPs commonly utilize nitrification and denitrification processes to biologically remove nitrogen from wastewater. Table 4-1 describes the pathways and conditions by which these processes occur.

Table 4-1: Denitrification and nitrification pathway description and formation of N_2O

Process	Pathway	Optimal Conditions for Microbes	N_2O formation	Conditions that Promote N_2O Formation
Denitrification (reduction of NO_3^-)	$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$	Anoxic: DO ideally less than 0.2 mg/L	Reduction of NO to N_2, with N_2O intermediary	High DO (1)
		pH: optimum is 7.0-8.5		Low pH (2)
		Temperature: 5-30C		High Nitrite (1)
		Organic carbon required		Low COD/N ratio (1)
Nitrification (oxidation of NH_3)	$NH_3 \rightarrow NH_2OH \rightarrow NOH \rightarrow NO \rightarrow NO_2^- \rightarrow NO_3^-$	Aerobic: DO ideally greater than 1 mg/L	From oxidation of NH_2OH or reduction of NO_2^-	Low DO (1)
		pH: optimum is 7.5 to 8.5		
		Temperature: 10-40C		High Nitrite (1)
		Low food to organism ratio		

(1): Kampschreur, Temmink et al. 2009
(2): Thorn and Sorensson 1996

In the denitrification step in WWTPs, N_2O formation increases with high dissolved oxygen, high nitrite concentration, and low COD/N ratio (Kampschreur, Temmink et al. 2009). All three of these factors are linked to decreased denitrification rates and lead to reduced N_2O emission. Wastewater pH of less than 6.8 is also expected to increase N_2O formation (Thorn and Sorensson 1996), increasing the yield of N_2O, i.e. the percentage of N_2 that is N_2O. This ratio of denitrified nitrate that becomes N_2O as compared to that which becomes N_2 partially determines the amount of N_2O emitted from the denitrification process (Beaulieu, Tank et al. 2011).

In the nitrification step (aerobic oxidation), low dissolved oxygen causes local oxygen limitation and an increase in a nitrifier denitrification pathway, in which nitrifying bacteria oxidize ammonia (NH_3) to nitrite (NO_2^-), and subsequently reduce NO to N_2O to N_2 (Wrage 2003; Kampschreur, Temmink et al. 2009). High nitrite accumulation leads to N_2O emissions due to the same nitrifier denitrification pathway.

After wastewater is treated in a WWTP, in the case that effluent is subsequently reused in urban agriculture, there are few models describing the fate of nitrogen. According to IPCC guidelines, nitrogen released to surface water is assumed to be converted to N_2O at an emission rate of 0.005 (range 0.0005-0.25) kg N_2O-N/kg N. Limited field data and assumptions about nitrification and denitrification in riverine systems were used to determine this emission factor. These assumptions are that all of the nitrogen is discharged with the wastewater and that N_2O production in the riverine system is directly related to nitrification and denitrification of the nitrogen in the wastewater (IPCC 2006). More recently, Beaulieu et al. carried out extensive experiments in US rivers to find that 0.75% of dissolved inorganic nitrogen inputs to riverine systems was converted to N_2O emissions by a combination of denitrification and nitrification (Beaulieu, Tank et al. 2011).

For soil application, the IPCC gives default emission factors for N_2O emissions from managed soils. These range from 0.003 kg N_2O-N/kg N for flooded rice fields to 16 kg N_2O-N/kg N for tropical organic crop and grassland soils (IPCC 2006). Wastewater reuse in cropped soils is not included. Understanding the fate of nitrogen in wastewater reuse for urban agriculture will also help in understanding the positive impacts of avoided fertilizer.

Therefore, a WWTP LCA using reported WWTP operations data along with modeling impacts external to WWTP is needed for a specific site in India. The objective of this study is to carry out a full systems life cycle assessment of Nallacheruvu WWTP in Hyderabad, India, including WWTP process emissions, end-use energy, embodied energy, and consequential emissions from off-site N_2O and CH_4 when WWTP effluent is reused in urban agriculture.

4.2 Wastewater Treatment Plant Description

The Nallacheruvu Wastewater Treatment Plant (NWWTP) was commissioned by the HMWSSB in 2007 after which it was inaugurated on February 25, 2009. The influent wastewater to this plant is thought to be domestic and it originates from open drains and closed pipes that are connected to the sewer network. It is difficult to estimate the service area and households served by NWWTP. Because NWWTP is estimated to treat 5% of the total wastewater treated in Hyderabad (table 3-2), the total area covered by the sewer network (294 km^2) was scaled down proportionally (Ministry of Urban Development 2010). From this calculation, the NWWTP serves approximately 15 square kilometers in and around the area of Uppal, Hyderabad. The total amount of properties in Hyderabad that are connected to sewers (551,026) (Ministry of Urban Development 2010) was also scaled down by the 5% proportion to result in 22,796 properties that are served by NWWTP. To estimate the amount of people served by NWWTP, the total population (7,597,058) was divided by total properties (2,028,435) in Hyderabad (Ministry of Urban Development 2010). Then, the amount of properties that are served by NWWTP was multiplied by 3.75 people/property, resulting in 85,377 people. The maximum capacity that can be handled by the NWWTP is 8 million gallons (US) per day, but actual treatment was as low as 3 MGD in the dry

Figure 4-1: Entrance to NWWTP

season (March-May 2010) and the average annual flow was 5 MGD for the first year of operation.

4.2.1 Physical Description

The processes at the NWWTP are in the following order (from inflow to outflow): coarse screen channel (20mm), pumped to inlet chamber, fine screen channel (6mm), detritor tank (settling and grit removal), upflow anaerobic sludge blanket reactors (UASB), facultative aerated lagoon, polishing pond, chlorination (as needed), and sludge drying. Biogas is collected at the UASB, scrubbed for H_2S removal, and flared. All data shown in figure 4-2 were provided by NWWTP authorities, except the change in chemical oxygen demand (COD), biochemical oxygen demand (BOD), and total organic carbon (TOC) across the UASB, which were independently measured by the researcher.

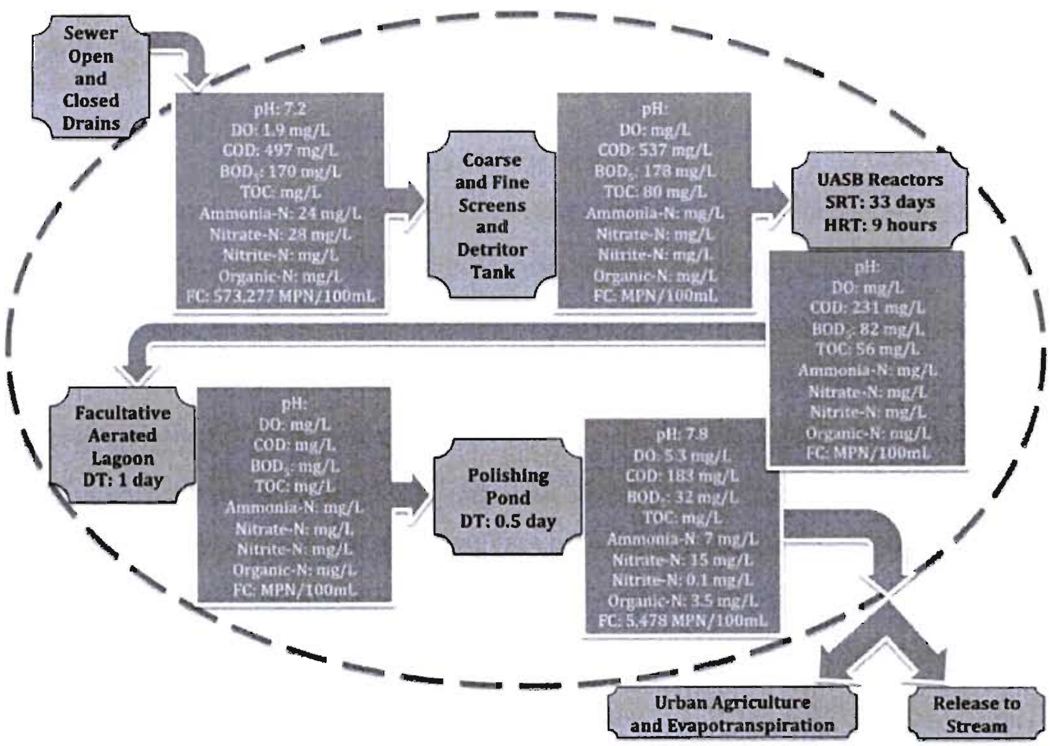


Figure 4-2: Process flow diagram of Nallacheruvu WWTP. The missing data has been requested from NWWTP. (SRT: solids retention time; HRT: hydraulic retention time; DT: detention time)

4.2.2 Energy Recovery

Biogas is produced and captured at the UASB. At full capacity, this plant is expected to generate 512 cubic meters biogas/day, which is expected to contain 60-65% methane (310 cubic meters methane/day) (Kumar 2010). In the future, power will be generated from a dual fuel genset. The expected power generation at full capacity of the plant is 705.6 kWh/day (Kumar 2010). Currently, power generation has not been started. Fugitive emissions of biogas from the UASB have not been measured.

Sludge is currently being generated and dried at the plant. At full capacity, 4 cubic meters grit/day, 12.6 cubic meters wet sludge/day and 6.28 metric tons dried sludge/day is expected (Kumar 2010). In the first year, only about 1 mt dried sludge/day was being generated (Kumar 2010). The plant had not yet decided the fate of the dried sludge, but it is dried and held on-site.

4.2.3 Process Insight

The influent wastewater to the NWWTP can be compared to the typical composition of high strength (60 gal/capita/day) untreated domestic wastewater, given by Metcalf and Eddy (Table 4-2).

Table 4-2: Typical composition of high strength untreated domestic wastewater (Metcalf and Eddy 2003) compared to influent wastewater to NWWTP reported from March 2009- March 2010 (Kumar 2010). TBD=to be determined.

Parameter	Untreated Domestic Wastewater	Influent WW to NWWTP
COD	800	497
BOD_5	350	170
TOC	260	TBD
Total Nitrogen	70	TBD
Ammonia	45	24
Nitrate	0	28
Nitrite	0	TBD
Organic	25	TBD
Fecal Coliform	10^5-10^8	$6*10^5$
TSS	85	353
VSS	315	128

4.2.3.1 Sewerage System

The dissolved oxygen in the influent wastewater to NWWTP (1.9 mg/L) suggests that the sewerage system is an aerobic environment. This is expected as the network of open drains and closed pipes is cascading, turbulent flow, which introduces oxygen into the wastewater. The high nitrate (NO_3^-) in this influent water may result from a combination of: nitrification occurring in the aerobic environment within the sewerage system, runoff from agriculture, and industrial effluents. Some nitrification is evident in the lower amount of ammonium (NH_4^+) and higher amount of NO_3^- (table 4-2), and is expected due to the optimal pH, DO, and temperature (table 4-1). However, the short retention time may hinder a large amount of nitrification.

Ammonification, or the conversion of organic nitrogen (NH_2) to NH_3 or NH_4^+, is also occurring within sewers. Biodegradable soluble organic nitrogen, a major component (90-95%) of total organic nitrogen, is converted to NH_4^+ when the pH is lower than the pKa (9.25), as is seen in NWWTP influent. At pH higher than pKa, biodegradable soluble organic nitrogen is expected to be converted to NH_3 and lost as a gas (Metcalf and Eddy 2003).

4.2.3.2 Screens and Detritor Tank

The screens remove some solids and provide more aeration. The settling in the detritor tank removes more solids and insoluble particulate organic nitrogen. The DO is expected to decrease here and the water is expected to start approaching an anoxic environment. Also, settling is expected to remove some portion of fecal coliforms and nematode eggs (George, Crop et al. 2002).

4.2.3.3 Upflow Anaerobic Sludge Blanket

Microbes in the wastewater use organic carbon as their food source and prefer to use oxygen as an electron acceptor until the DO is below 0.2 mg/L. Then, the microbes are forced to use NO_3^-) as an electron acceptor. NO_3^- is expected to be reduced as shown in table 4-1. Denitrification causes the pH to increase, and may result in a deprotonation of NH_4^+ to NH_3 and resulting loss of NH_3 gas. N_2O yield (percentage of denitrification) is expected to decrease to undetectable levels (~0) when pH is higher than 6.5 to 7 (Thorn and Sorensson 1996). Finally, organic nitrogen is not expected to change much due to UASB processes (Arceivala and Asolekar 2007)

4.2.3.4 Facultative Aerated Lagoon

The aerobic environment allows for more extensive nitrification as conditions are optimal for nitrifiers to oxidize NH_4^+ to NO_3^-. In the earlier part of the basin, heterotrophic organisms feed on organic matter and consume oxygen, while nitrifiers grow in the later part where little organic matter is present and there is less competition for oxygen. Nitrite is not expected to accumulate at any point in the WWTP, and if it did, it would mean that toxic conditions have killed the necessary microbes (Novotny 2006). BOD, COD, and TOC should all decrease due to the cascading aeration. However, at NWWTP, the detention time of 1 day may be too low for significant nitrification. We have requested two sets of in-plant measurements to verify this.

4.2.3.5 Polishing Pond

The final step for NWWTP is often the polishing pond. Removal of additional BOD, COD, TOC, nutrients as well as fecal coliforms and nematode eggs is expected here through settling (Spellman 2009). A chlorination bed follows but is not often used.

4.2.4 NWWTP Performance Data from HMWSSB

The performance data provided by the Hyderabad Municipal Water Supply and Sewerage Board was averaged for one year (March 2009-March 2010) and is shown in table 4-3.

Table 4-3: NWWTP measured parameters and treatment efficiencies for March 2009- March 2010 (Kumar 2010)

Parameter (Units)	Average Influent to NWWTP	Average Effluent from NWWTP	Removal % for Pollutants	Disposal Standards in appropriate units
	Average ± Standard Deviation (n= # of data points)			
pH	7.2 ± 0.7 (n=46)	7.8 ± 0.9 (n=46)	---	5.5-9.0
DO (mg/L)	1.9 ± 2.1 (n=44)	5.3 ± 2.4 (n=44)	---	---
BOD5 (mg/L)	170 ± 87 (n=46)	32 ± 43 (n=46)	81%	30
COD (mg/L)	497 ± 207 (n=45)	183 ± 189 (n=45)	62%	250
TSS (mg/L)	353 ± 251 (n=27)	23 ± 43 (n=27)	93%	100
VSS (mg/L)	128 ± 77 (n=24)	8.1 ±17 (n=24)	94%	---
Fecal Coliforms (MPN/100mL)	573,277 ± 300,837 (n=26)	5,478 ± 2,206 (n=26)	99%	10,000

The average effluent discharges at NWWTP are reported to meet disposal standards set by the Indian Central Pollution Control Board. Chlorination is available at NWWTP, but it is rarely used because the effluent fecal coliform concentrations meet disposal standards. There is evidence that chlorination produces N_2O during decompostition of monochloramine (NH_2Cl) at neutral pH (Hashimoto 1981). Because chlorination is not often utilized in this WWTP, these emissions are thought to be negligible.

4.2.5 UASB WWTP Performance

The NWWTP performance is in line with the literature. Concerning pathogen treatment efficiency, UASB paired with a polishing unit alone can remove 99.8% nematode eggs (Tyagi, Sahoo et al. 2010). UASB technology is useful in meeting the maximum permissible limits of fecal coliform for disposal (10,000 MPN/100mL), but the suggested desirable limit

Figure 4-3: UASB with biogas capture (pipes at top) at NWWTP

of 1,000 MPN/100mL is not often met (Khalil, Sinha et al. 2008; Tare and Nema n.d.). Therefore, tertiary treatment is needed and chlorination is often planned, but it is only used to meet the maximum permissible limits.

In the case of resource recovery, which was a major attraction in choosing UASB implementation, many plants that planned to utilize biogas for electricity generation have not yet started and sludge is not yet being utilized in a significant way (Kumar 2010; Tare and Nema n.d.). For power generation, this delay could be for many reasons, including low biogas generation in small and medium sized WWTPs and in cold months in Northern India. Also, there is very little incentive for WWTP to start energy recovery from biogas for the following reasons: the UASB WWTP has a low energy requirement, power outages do not greatly affect the technology, the energy bill is linked to the installed load of the WWTP, and there are upfront costs and risks to starting and maintaining power generation from biogas (Tare and Nema n.d.). Utilizing dried sludge has also been problematic as there is not a reliable or lucrative market for the sale of sludge (Kumar 2010; Tare and Nema n.d.).

4.2.6 Consequential Life Cycle Assessment

Consequential LCA is used to determine how flows and related impacts will change as a result of decisions and actions taken outside the WWTP boundary. Consequential LCA has been used to quantify GHG impacts due to changes in flows in response to policy decisions, such as the ramping up of corn-based biofuels in the US (Zhang, Spatari et al. 2010). At a time when some studies (Farrell, Plevin et al. 2006; Argonne National Laboratory 2008) suggested that biofuels could reduce atmospheric GHGs as growing feedstock sequesters carbon, Searchinger et al. used consequential LCA to determine that GHGs may actually increase as a result of land use change. This group modeled land use change due to a worldwide farmer response to growing demand and prices for biofuels, and predicted that they would subsequently convert grasslands, forests, etc, to cropland (Searchinger, Heimlich et al. 2008). Consequently, they modeled GHGs to double in the coming years.

In this WWTP LCA, decisions about both in-boundary and out-of-boundary flows will be considered. Because the WWTP does not have direct control over what happens to the released water after treatment, GHG impacts from WWTP effluent when it is released to the environment or reused in urban agriculture is considered out-of-boundary, and could have land use change implications. Two other consequential impacts will be included: avoided fertilizer due to the nutrient content of WWTP effluent reused in urban agriculture, and avoided electricity from the grid when biogas is reused to generate electricity.

4.3 Life Cycle Assessment Methodology

The following equation (4 components) is used to quantify total on-site life-cycle energy use and GHG emissions. The four components include:

- End-use energy in WWTP operations,
- Process emissions of methane (CH_4),
- Process emissions of nitrous oxide (N_2O), and
- Embodied energy of infrastructure.

$$LCGHG_{Annual} = \sum_{n} [(E_{On-site} * EF_{Electricity}) + (W_{CH_4\text{-}UASB_{Fugitive}} * GWP_{CH_4}) + (W_{CH_4\text{-}Captured_{Leak}} * GWP_{CH_4})$$
$$+ (W_{N_2O} * GWP_{N_2O}) + ((W_{Cement} * EF_{Cement}) / T_{Lifetime})]$$

(Equation 4-1)

where: $E_{On\text{-}site}$= on-site energy use at the WWTP (only electricity);

$EF_{Electricity}$= average CO_2 emission factor from thirteen thermal power plants (Coal, Oil, and Gas Fuels) in the State of Andhra Pradesh (2008-2009);

$W_{CH4\text{-}UASBFugitive}$= kg methane lost from UASB as fugitive emissions per year;

GWP_{CH4}= global warming potential of methane: 24 $kgCO_2e/kgCH_4$;

$W_{CH4\text{-}CapturedLeak}$= kg methane leaked after being captured per year;

W_{Cement}= total kg cement used in WWTP infrastructure;

EF_{Cement}= CO_2 emission factor for cement manufacture;

$T_{Lifetime}$= average lifetime of WWTP infrastructure: 30 years;

W_{N2O}= kg nitrous oxide emitted from nitrification and denitrification processes on-site at NWWTP per year;

GWP_{N2O}= global warming potential of nitrous oxide: 298 $kgCO_2e/kgN_2O$.

Equation 4-1 is applied to assess energy and GHG emissions for LCA of WWTP in India with emphasis on UASB technology.

For the WWTP LCA, scope 1 emissions come directly from the wastewater in the form of N_2O and CH_4, scope 2 emissions come from end-use electricity, and scope 3 emissions are the embodied energy of infrastructure. Consequential LCA addresses the full system inside and outside the WWTP boundary comparing various scenarios with and without wastewater agriculture, and will be discussed in a separate section. This consequential LCA is important, as little is known about life cycle GHG impacts of WWTP when the wastewater is subsequently used for agriculture.

Greenhouse gas impacts of WWTPs are not well characterized in developing countries. The IPCC Guidelines for National Greenhouse Gas Inventories provides a method to calculate GHG emissions from untreated wastewater for India (Doorn, Towprayoon et al. 2006), although is has high uncertainty. When WWTPs are built in developing countries, there is an expected increase in energy use and greenhouse gas emissions from WWTP electricity use. However, if CH_4 capture is used in the WWTP, significant GHG emissions can be mitigated. These consequential LCA impacts will be quantified in section 4.4.2, following WWTP LCA.

4.3.1 Goal and Scope of WWTP LCA

LCA goal and scope definition serve the purpose of setting the boundaries of the project and help to determine the most appropriate functional unit. The scope for this LCA includes WWTP process boundaries, with consequential LCA representing various scenarios outside the boundary. The functional unit used was million gallons wastewater treated per year (sometimes expressed as per gallon since this LCA was done as NWWTP had been operating for one year).

4.3.2 WWTP LCA Data

4.3.2.1 Scope 1 Direct Emissions from WWTP Processes

4.3.2.1.1 Fugitive emissions of methane from UASB

Even though methane capture is an integral part of the UASB, fugitive emissions from the water surface are expected. Biogas yield can be estimated from the amount of COD removed during the treatment process. This theoretical biogas yield is between 0.35 and 0.5 cubic meters biogas/kg of COD removed (IPCC 2006; Tare and Nema n.d.). However, for UASB technology the actual yield is expected to be only 25-30% of this value or 0.08-0.1 cubic meters biogas/kg of COD removed (Tare and Nema n.d.). The majority of the biogas remains dissolved in the effluent, and increases its BOD and COD. It is assumed that this biogas is made up 65% methane and 32% carbon dioxide by volume (Monteith, Sahely et al. 2005; Kumar 2010); 0.651 kgCH4/cubic meter CH4 was used for the density of methane (at 1atm and the average daily temperature of Hyderabad, 27.4°C (Rao 2010)). The difference between the amount produced and the amount biogas captured was then calculated and assumed to be lost to the environment.

$$W_{CH_4\text{-}UASB_{Fugitive}} = \frac{(V_{Biogas\text{-}Yield} - V_{Biogas\text{-}Captured}) * P_{CH_4} * D_{CH_4}}{MG} \qquad \text{(Equation 4-2)}$$

where: $V_{Biogas\text{-}Yield}$= cubic meters of theoretical biogas yield from the UASB per year, based on COD reduction;
$V_{Biogas\text{-}Captured}$= cubic meters of biogas captured from UASB per year;
P_{CH4}= proportion of biogas that is methane: 0.65 cubic meters CH4/cubic meters biogas;
D_{CH4}= density of methane: 0.651 kg CH_4/cubic meter CH_4 at average daily temperature for Hyderabad (27.4°C);
MG= million gallons: 1,736 million gallons wastewater treated per year.

4.3.2.1.2 Methane leakage after capture

Because the volume of biogas currently captured is not measured, the direct emissions from biogas were estimated as a proportion of that expected for full capacity of the plant. Even though biogas is flared at this plant, incomplete combustion and leaks are expected and a 5% undetected biogas leak rate can be assumed (Sahely, MacLean et al. 2006).

$$W_{CH_4\text{-}Captured_{Leak}} = \frac{V_{Biogas\text{-}Captured} * R_{Leak} * P_{CH_4} * D_{CH_4}}{MG} \qquad \text{(Equation 4-3)}$$

where: R_{Leak}= undetected biogas leak rate due to incomplete combustion and leakage: 5% according to Sahely et al. 2006.

4.3.2.1.3 Fugitive emissions of nitrous oxide from UASB and Oxidation Pond

Few people have measured N_2O and CH_4 emissions from UASB and oxidation ponds. Although it is thought that denitrification in WWTP anoxic zones is the largest contributor to N_2O emissions (US EPA 2009), nitrification in WWTP aerobic zones has been found to be significant, especially in WWTP where both anoxic and aerobic processes are used (Kampschreur, van der Star et al. 2008; Ahn, Kim et al. 2010). Kampschreur et al 2009 completed a comprehensive review of many different types of WWTP processes and their related emissions (Kampschreur, Temmink et al. 2009) and Ahn et al measured emissions from many different technologies.

For this WWTP LCA, fugitive emissions from the anaerobic UASB reactors and the aerobic facultative aerated (oxidation) pond were estimated from similar technologies described in Ahn et al: the anaerobic portion of a separated biological nutrient removal (BNR) WWTP at approximately 0.001% as kgN2O-N/kg total nitrogen removed (10% of 0.01%), and an oxidation ditch at 0.03% kgN_2O-N/kg total nitrogen removed (Ahn, Kim et al. 2010). Therefore, nitrous oxide emissions were estimated at 0.031% (range 0.02%-0.04%) as kgN_2O-N/kg total nitrogen removed. Ahn et al also showed the % of influent TKN that becomes N_2O. In order to make comparisons with the percent of influent nitrogen in IPCC methodology, this is the number shown in table 4.4.

Table 4-4: N_2O emissions from WWTP processes: a comparison of results using IPCC methodology to findings by Ahn et al 2010.

Source	Method	Equation	mg N2O/ person/ year (Range)	EF: % as kg N2O-N/kg influent N (range)
IPCC 2006	Calculated for Hyderabad, India	N_2O_{WWTP}= Population (HYD) *degree of utilization of modern, centralized WWTP % (used % ww collected in HYD) * fraction of industrial and commercial co-discharged protein (default 1.25 given by IPCC) * EF kgN_2O/person/year (0.0036)	628 (201-2,715)	0.02 (0.01-0.04)
Ahn et al. 2010	Whole Plant	Researchers in the US measured mass flux from each zone in the WWTP and normalized to the daily influent total Kjeldahl nitrogen (TKN) loading	280 (150-410)	0.01
	Anaerobic portion of BNR Process		28 (15-41)	0.001
	Aerobic Oxidation Ditch		1800 (1030-2570)	0.03

4.3.2.1.4 Scope 1 Direct Emissions from On-site WWTP Operations

Emissions from vehicle transport operations are very low because there are no municipally-owned or company-owned cars specifically for WWTP use.

Motorbikes are used by a few workers, but mainly for transport to and from work, and not often for transport around the WWTP grounds. Human-powered pushcarts are the most common form of transport around the WWTP. Electricity was the major form of energy used at the Nallacheruvu WWTP; diesel and natural gas use were reported to be very little to none (Kumar 2010). Therefore, emissions from gasoline, diesel, and natural gas are not included.

4.3.2.2 Scope 2 Emissions from Electricity Use for WWTP Operations

The actual on-site electricity use for the plant was 3.3 MWh/day for the first year of operation, equivalent to 0.7Wh/gal (6.2 MWh/day is expected at full capacity). An average of emission factors for 13 Thermal Power Plants in the state of Andhra Pradesh (Coal, Oil, and Gas Fuels) (2008-2009) is 0.747±0.3 $mtCO_2/MWh$ (only carbon dioxide emissions were reported here) (India Central Electricity Authority 2009).

4.3.2.3 Scope 3 Emissions from Embodied Energy in WWTP Materials

On-site building materials are dominated by concrete use for construction of the WWTP. Concrete is used for: coarse screen channel, main pumping station, inlet chambers, fine screen channel, detritor tank, division box 1&2, distribution box, UASB reactors, facultative aerated lagoon, polishing pond, chlorine mixing tank, chlorine contact tank, sludge pump house, sludge drying beds, gas holder, gas scrubber/blower room, biogas genset room, biogas flare unit, chlorination room, mechanical, electrical and plumbing room, and administration block house. On-site piping includes: raising main (one cast iron), distribution lines (two cast iron and one high density polyethylene), sludge lines (two cast iron), filtrate line (one stainless steel), and gas line (one fiberglass reinforced plastic).

Because incomplete data was provided for piping infrastructure, the infrastructure material emissions are calculated only for concrete use. An estimated total 17,022 metric tones of concrete was calculated from widths and lengths provided by Mr. Kumar of NWWTP, and some heights were estimated by this researcher. An emission factor of 0.33 mt CO_2e/mt concrete (Pitterle 2008) was used. A typical lifetime of a WWTP's pumps, tanks, and other technical parts is fifteen years while buildings, filter beds and pipes is 30 years (Lundin, Bengtsson et al. 2000; Foley, Haas et al. 2005). Therefore, the emissions from cement were divided by 30 years.

4.3.2.4 Consequential Evaluation of GHG Emissions

N_2O and CH_4 emissions from WWTP effluent released to both urban agriculture plots and the riverine system were quantified as out-of-boundary emissions. For urban agriculture, these emissions can be quantified with greater certainty because N_2O emissions from managed soils is well-studied (Del Grosso, Ojima et al. 2009) and CH_4 is expected to be very low due to the aerobic environment of land application. However, these emissions from rivers and streams are more uncertain.

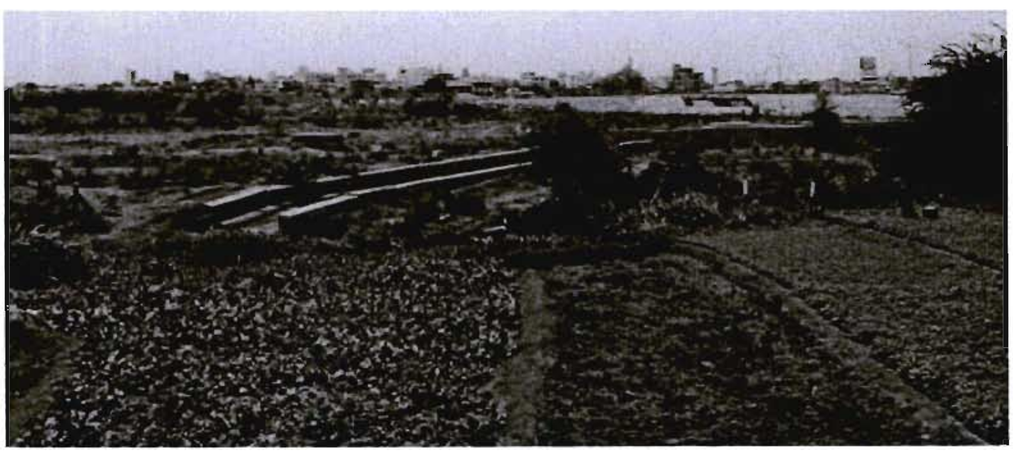


Figure 4-4: NWWTP effluent discharge to the stream, with urban agriculture plots nearby

The impacts of two main scenarios were evaluated:

- Uncontrolled release of wastewater without treatment and

- Treatment at NWWTP with partial reuse in urban agriculture with the remainder flowing into the riverine system downstream of NWWTP.

For NWWTP treated effluent release to urban agriculture, the amount of land adjacent to the WWTP was estimated using Google Earth's ruler tool. The amount of water that would be used by the total area over 10 months (agriculture was estimated to stop for 2 months per year due to monsoon rains as per interviews with local farmers) was estimated from the volumes used in this report's site study (see Chapter 5).

In addition to the two core scenarios, avoided GHGs were computed from:

- Avoided fertilizer and
- Avoided electricity due to the reuse of biogas.

Avoided GHG emissions from avoided fertilizer and electricity from biogas reuse were calculated in the following ways. For urban agriculture, avoided emissions from fertilizer use that wasn't needed due to the nutrient content of treated wastewater were credited to the off-site N_2O emissions. Concentrations of total soluble nitrogen, phosphorus, and potassium (N, P, K), the major components of synthetic fertilizers, in treated effluent water were measured at the site (see chapter 5). These amounts were scaled up for use in the adjacent area to NWWTP over 10 months per year. Then, to determine the avoided GHG emissions from not using synthetic N, P, and K fertilizers, the amounts were multiplied by emission factors compiled by Pitterle and totaled (Pitterle 2008).

$$GHG_{Avoided_{Fertilizer}} = (C_{IrrigationWater_{Nitrogen}} * EF_{Fertilizer_{Nitrogen}}) + (C_{IrrigationWater_{Phosphorus}} * EF_{Fertilizer_{Phosphorus}}) + (C_{IrrigationWater_{Potassium}} * EF_{Fertilizer_{Potassium}})$$

(Equation 4-4)

where: $GHG_{Avoided\ Fertilzer}$= avoided CO_2 equivalent emissions from fertilizer;
$C_{Irrigation\ Water\ (N,\ P,\ K)}$ = concentration of nutrients in irrigation water

(based on treated effluent water measurements from Chapter 5);

$EF_{Fertilizer\,(N,\,P,\,K)}$ = emission factor: 4.57 kg CO_2e/kg-N, 1.25 kgCO_2e/kg-P, or 1.29 kg CO_2e/kg-K fertilizer (Pitterle 2008).

These avoided emissions were then credited to the off-site N_2O emissions or N_2O from agriculture in the results.

As stated earlier, biogas that is currently flared on-site at NWWTP could be used to generate electricity. This potential electricity credit was calculated by using a net calorific value of 23.3 MJ/m^3 for biogas with 65% methane from anaerobic digestion in a WWTP (Bonnier 2008). A 25% power plant efficiency for biogas to electricity from Pitterle's work at a US WWTP is used here (Pitterle 2008).

$$GHG_{Avoided_{Electricity}} = V_{Biogas-Captured} * NCV_{Biogas} * \eta_{pp-Biogas} * X_{kWh-MJ} * EF_{Electricity}$$

(Equation 4-5)

where: $GHG_{Avoided\ Electricity}$= avoided CO_2 equivalent emissions from electricity generated from biogas;

NCV_{Biogas}= net calorific value of biogas: 23.3 MJ/m^3 (Bonnier 2008);

$\eta_{pp-Biogas}$= US average power plant efficiency: 25% (Pitterle 2008);

X_{kWh-MJ}= conversion: 0.2778 kWh/MJ;

$EF_{Electricity}$= average CO_2 emission factor from thirteen thermal power plants (Coal, Oil, and Gas Fuels) in the State of Andhra Pradesh (2008-2009) (India Central Electricity Authority 2009).

This result was credited to on-site WWTP emissions.

Finally, to yield total emissions per million gallons, these avoided emissions were divided by 1,736 million gallons of wastewater treated by NWWTP per year, then multiplied by the amount of water appropriate for each scenario in the results.

4.3.2.4.1 Methods for the Two Core Scenarios

4.3.2.4.1.1 Untreated Base Case

GHG emissions from direct release of untreated wastewater was estimated using IPCC methodology for both CH_4 and N_2O releases to streams (IPCC 2006). For methane, the IPCC methodology and variation was used and is described in Equation 4-6.

$$GHG_{Riverine_{CH_4}} = C_{Influent-COD} * (B_o * MCF) * GWP_{CH_4} \qquad \text{(Equation 4-6)}$$

where: $GHG_{Riverine\ CH4}$= CO_2 equivalent emissions from methane released from riverine systems per million gallons wastewater treated;

$C_{Influent-COD}$= concentration of COD in influent wastewater: 1,881kg COD/million gallons (Kumar 2010);

B_o= maximum CH_4 producing capacity: 0.25 $kgCH_4/kgCOD$ (IPCC 2006);

MCF= methane correction factor for rivers and lakes, an indicator on the degree of which the system in anaerobic: 0.1 (IPCC 2006);

GWP_{CH4}= global warming potential of nitrous oxide: 24 $kgCO_2e/kgCH_4$.

The variation CH_4 emissions were calculated from the variation in MCF, which ranged from 0 to 0.2 for rivers and lakes (IPCC 2006). In IPCC methodology, the emission factor for CH_4 from riverine systems is given by B_o*MCF.

To estimate N_2O emissions from wastewater in riverine systems, IPCC methodology along with a PNAS study was used. IPCC assumptions result in the estimation that N_2O emissions from nitrification are double those from denitrification in streams (Mosier, Kroeze et al. 1998; Beaulieu, Tank et al. 2011). In a PNAS study by Beaulieu et al. 2011, researchers carried out an extensive study in order to improve the estimate of the total amount of nitrogen that is converted to N_2O in streams.

Their results showed no correlation to river network length, even though longer water residence times were expected to increase the percentage of N being denitrifed. Regardless, the global range was 0% to 0.9% conversion of N inputs to N2O via denitrification (figure 4-5).

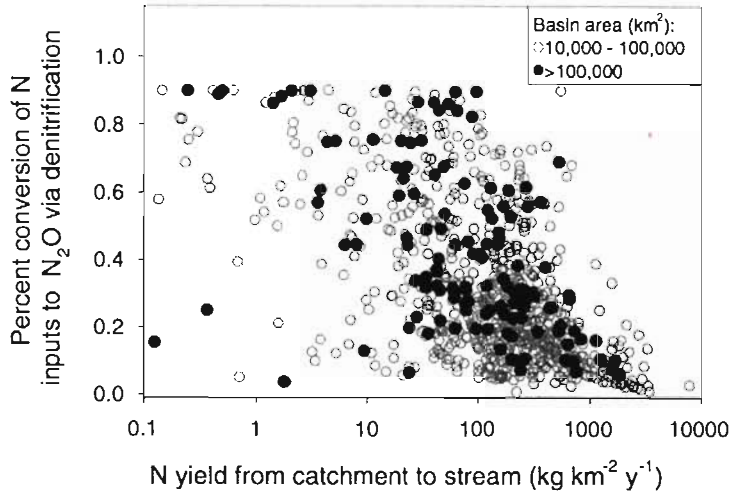


Figure 4-5: Results from Beaulieu et al 2011. Percent conversion of N inputs to N_2O via denitrification versus the nitrogen yield from catchment to stream (indicator of river network length) of 866 rivers worldwide

Ultimately, Beaulieu et al. estimates that the percentage of dissolved inorganic nitrogen was converted to N_2O via denitrification and nitrification in rivers is 0.75% (Beaulieu, Tank et al. 2011). For this WWTP consequential LCA, the average value reported was using the IPCC emission factor ($0.005 kgN_2O/kgN$), and shown in equation 4-7.

$$GHG_{Riverine_{N_2O}} = C_{InfluentNitrogen} * EF_{Riverine_{N_2O}} * GWP_{N_2O} \quad \text{(Equation 4-7)}$$

where: $GHG_{Riverine\ N2O}$= CO_2 equivalent emissions from nitrous oxide released from riverine systems per million gallons wastewater treated;

$C_{Influent\text{-}Nitrogen}$= concentration of inorganic nitrogen in influent wastewater to NWWTP (Kumar 2010);

$EF_{RiverineN2O}$= the default IPCC emission factor for N_2O emissions from domestic wastewater nitrogen effluent from nitrification and denitrification in rivers and estuaries: 0.005 kgN_2O/kgN (IPCC 2006);

GWP_{N2O}= global warming potential of nitrous oxide: 298 $kgCO_2e/kgN_2O$.

To calculate the variation in N_2O emissions from riverine nitrogen, the emission factor of 0.0075 kgN_2O/kg dissolved inorganic nitrogen from the PNAS article was used and the range is shown in Figure 4-11.

These N_2O and CH_4 emissions per gallon of untreated wastewater are multiplied by the amount of water appropriate for each scenario in the results.

4.3.2.4.1.2 DAYCENT for estimating of N_2O emissions from agriculture

Because few people have measured N_2O directly from urban agriculture with wastewater irrigation, this researcher had planned to measure N_2O emissions in the site study (chapter 5). However, the equipment in India did not have the necessary parts, and permission to use equipment in the US was not given due to instrument contamination associated with these studies. Therefore, the DAYCENT model was used.

DAYCENT, developed by a group at the Natural Resource Ecology Laboratory at Colorado State University (CSU), is a well-documented and widely used model for estimating GHG emissions from cropped fields (Del Grosso, Mosier et al. 2005; Del Grosso, Ojima et al. 2009). It has been used and validated by researchers as well as the US Environmental Protection Agency (Jarecki, Parkin et al. 2007; US EPA 2011). It is most often used to estimate N_2O emissions for major crops (wheat, corn, soybeans, etc) with commercial fertilizer use. It has not been used for wastewater agriculture, for vegetables, or for India.

Inputs to the model included (and detailed in Appendix A):

- Weather specific to Hyderabad: obtained for 2000-2010 from Dr. Kesava Rao, a scientist of Agroclimatology at ICRISAT;
- Historical data: assumptions were made on agriculture frequency, type, nutrient delivery, etc from year 1 until this study started. It is known that

agriculture began in the area around the late 1960s and continued until the wastewater treatment plant was built. Grazing of buffaloes in the area has been occurring and fire is often used to clear grasses to start cultivation.

Figure 4-6: Buffaloes grazing at NWWTP.

- Soil characteristics: physical and chemical parameters that were determined from lab tests described in chapter 5;

- Crop characteristics: growth type, fraction of carbon allocated to roots, weather conditions appropriate for growth, etc. which were discussed with Dr. Parton at CSU. Also, grams carbon per kg of spinach was needed to determine net primary productivity (National Council on Radiation Protection and Measurements 1983).

- Nitrogen and organic matter delivered in the water were scaled from year 1 to 2011 from water nutrient and organic matter test results in the site study (for further description of sites and nutrients, see chapter 5). The seasonal distribution was based on measured amounts of riverine nitrogen at the basin month of the Ganges (Green, Vorosmarty et al. 2004). Then, based on the change in river nitrogen load from 1970-2010 (estimated at 50% for Brazil-Russia-India-China (BRIC)), nitrogen was scaled accordingly for the last 40 years (Bakkes, Bakkes et al. 2008). Previous nutrients were scaled linearly back to year 1.

- Fertilizer and organic matter addition events were scheduled along with irrigation events, as these components were delivered in the irrigation water.
- In the model, all study activity took place on the same days as they actually occurred. After the site study in March-April 2010, a monsoon season was simulated and cultivation began again in late June 2010 and continued for 9 additional identical growing cycles through February 2011. For yearly data, the emissions from these 10 growing cycles were summed. Input data are detailed in Appendix A.

Output N_2O emissions from the model were summed through the end of 2011, as fluxes were seen, and expected, for many months after the study ended. The first growing cycle was based identically on the actual field study. For the 9 additional growing cycles, modeled irrigation, nitrogen fertilizer, and organic

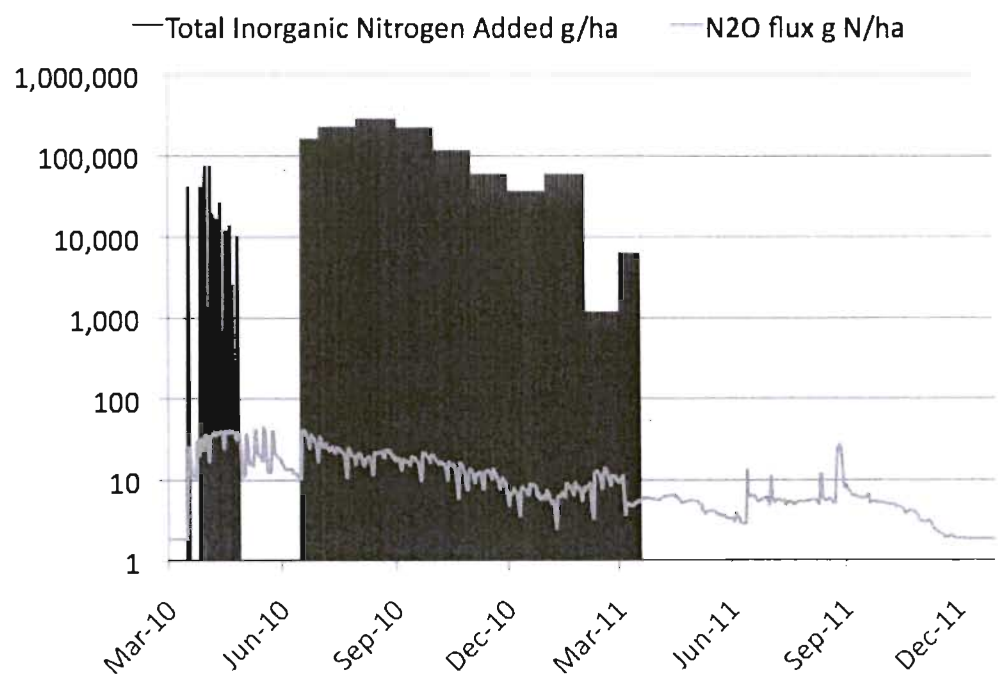


Figure 4-7: Output of total inorganic nitrogen added in treated effluent irrigation water as compared to nitrous oxide flux from 10 growing cycles of spinach-cultivated land

matter addition events occurred once every three days Fluxes of N_2O, that are higher than the baseline, can be seen for about 9 months after the last addition of nitrogen (figure 4-7).

In figure 4-7 a steady baseline concentration of 1.9gN/hectare for N_2O flux can be seen. This background was subtracted from the total N_2O flux in the consequential LCA results.

In the DAYCENT model, net primary productivity depends on a temperature curve representing the conditions for which the crop grows best and other growth parameters that were refined for the site irrigated with untreated water (chapter 5). The modeled output for productivity at the site irrigated with treated water was also similar to the actual measured productivity. However, the site irrigated with groundwater had higher modeled productivity than the actual. This is thought to be due to nutrients in lower soil levels that the model is including, but which the actual crop could not access (table 4-5) (Del Grosso, Parton et al. 2011). Background concentrations are not subtracted from values in table 4-5.

Table 4-5: Example of DAYCENT outputs using agricultural plots for one growing cycle only (chapter 5)

			Irrigation Water at Site		
Output	Units	Source	Groundwater	Treated Effluent	Untreated Surface Water
Net Primary Productivity	gC/ha	Actual	67	759	948
		DAYCENT Model	485	671	950
N2O nitrification	gN/ha		2,082	2,235	1,291
N2O denitrification	gN/ha		25	0	0
N2O flux	gN/ha		2,108	2,235	1,292
NO flux	gN/ha		3,821	11,399	9,766
N2 denitrification	gN/ha		16	0	0

For this WWTP consequential LCA, N_2O emissions per square meter from DAYCENT were divided by the gallons of water used per square meter per year (determined from water quantity used for the treated effluent plot in chapter 5), and multiplied by the amount of water that could be applied to agriculture for the different scenarios. N_2O is the only GHG considered here because the aerobic environment of agriculture oxidizes more CH_4 than it releases. Therefore, CH_4 emissions fall to zero when the treated wastewater is reused in agriculture.

4.4 LCA Results

4.4.1 On-site Energy Use and GHG Emissions

The in-boundary results of this study can be described by the following efficiency metrics for water use, energy use and related operating costs, and GHG emissions for the NWWTP.

4.4.1.1 Wastewater Generated

Wastewater collected for NWWTP is 56 gallons (US) per person per day as a yearly average. In 2006, India's average per capita wastewater generation was 36 gallons/day (United Nations Development Programme 2006). Because the sewer network is a combination of closed pipes and open drains, they are expected to collect stormwater during the rainy season and other water/effluents that are dumped by residents.

4.4.1.2 Energy Use

The actual electricity use at this WWTP was calculated for the average 5MGD (18MLD) over the first year of operations. The average electricity to treat one gallon of wastewater at this plant was 0.7Wh/gal, much less than that reported in the US due to UASB selection. Finally, the cost of electricity only for treating wastewater for the Nallacheruvu WWTP comes to approximately $0.0001 USD/gallon wastewater treated (or 0.004 INR/gallon wastewater treated).

4.4.1.3 GHG Emissions

The greenhouse gas emissions from this WWTP come from electricity use, infrastructure, CH_4 leakage before and after capture from the UASB, and N_2O emissions from the denitrification and nitrification processes in NWWTP. The total emissions from CH_4 were 435mgCO_2e/gallon wastewater treated and the total N_2O emissions were 9mgCO_2e/gallon wastewater treated as seen in table 4-6.

Table 4-6: On-site energy use and GHG emissions for NWWTP

<table>
<tr><th>Item</th><th colspan="3">Nominal (Average) Flow Parameter (A)</th><th colspan="2">Comparative Benchmark (USA)</th><th>Emission Factor or Global Warming Potential (B)</th><th>GHG Contribution (=A*B)</th></tr>
<tr><td>Operations Electricity</td><td colspan="3">0.7 Wh/gallon (1)</td><td colspan="2">2.4 Wh/gallon (5)</td><td>747 mgCO2e/Wh (6)</td><td>518 mgCO2e/ gallon</td></tr>
<tr><td rowspan="2">COD Reduction in UASB for Methane Production</td><td colspan="2">1,158 mg COD reduced/gallon (1)</td><td>17 mg CH_4 fugitive emissions/gallon (2,7)</td><td colspan="2" rowspan="2">211 mg CH4 captured/gallon (5)</td><td rowspan="2">24 mgCO_2e/ mgCH_4 (7)</td><td rowspan="2">435 mgCO2e/ gallon</td></tr>
<tr><td colspan="2">27 mg CH4 captured/gallon (1)</td><td>1.4 mg CH4 leaked from capture/gallon (3)</td></tr>
<tr><td rowspan="2">Nitrous Oxide Production as a Percentage of Total Nitrogen Removal</td><td rowspan="2">99 mgTN removed/ gallon (1)</td><td>Anaerobic Process</td><td>0.001% as mg N2O-N/mg N (4)</td><td>111 mgTN/ gallon (5)</td><td>0.077% mgN2O-N/mgN (5)</td><td rowspan="2">298 mgCO_2e/ mgN_2O (7)</td><td rowspan="2">9 mgCO2e/ gallon</td></tr>
<tr><td>Aerobic Process</td><td>0.03% (0.02-0.04) as mg N2O-N/mg N (4)</td><td colspan="2">103 mgCO2e/gallon for NWWTP (7)</td></tr>
<tr><td>Concrete</td><td colspan="3">327 mg concrete/annual gallon (1)</td><td colspan="2">163 mg concrete/ annual gallon (5)</td><td>0.33 mgCO2e/ mg concrete (5)</td><td>108 mgCO2e/ annual gallon</td></tr>
</table>

(1): Kumar 2010
(2): Tare and Nema n.d. See section 4.3.2.1.1
(3): Sahely, MacLean et al. 2006
(4): Ahn, Kim et al. 2010
(5): Pitterle 2008
(6): India Central Electricity Authority 2009
(7) IPCC 2006

The emissions per gallon of water treated for each item as well as the total on-site emissions are shown in figure 4-8.

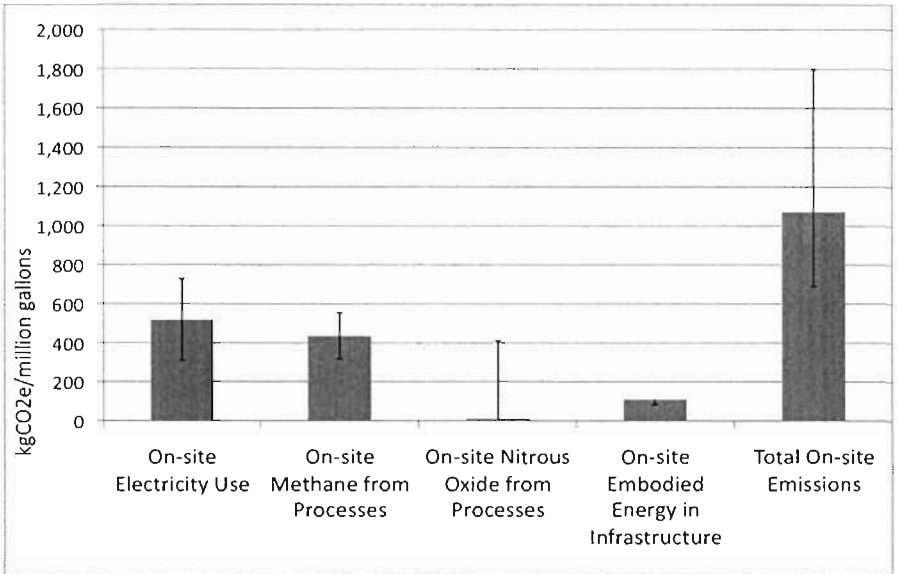


Figure 4-8: On-site energy-related and process GHG emissions at NWWTP. Error bars show the range in emissions.

The ranges shown for on-site electricity-related emissions are due to the range in the emission factor (standard deviation of 300mgCO2e/Wh) (India Central Electricity Authority 2009), for on-site methane from processes are due to the range in COD conversion to biogas (0.08-0.1 m^3/kg COD removed) (Tare and Nema n.d.), for on-site nitrous oxide from processes are due to the range in total nitrogen converted to N_2O in aerobic processes (Ahn, Kim et al. 2010), and for embodied energy in infrastructure are due to the range in the emission factor (0.25-0.33 mgCO_2e/mg reinforced concrete) (Pitterle 2008).

4.4.1.4 Limitations of WWTP Data

While the on-site energy use from NWWTP (0.7Wh/gal) is within the range for Indian cities shown in Chapter 2 (0.2-5Wh/gal), there was no way to verify this data independently. Data reliability is an ongoing issue that is found in many LCA studies and is acknowledged here. As mentioned previously, further measurements of flows throughout NWWTP (figure 4-2 and table 4-2) have been requested and are currently being measured. However, the results have not been obtained at the time of this publication.

4.4.2 Consequential LCA Results

Figure 4-10 shows the results from various modeled efforts towards reducing greenhouse gas impacts of wastewater. In this case study, when treated wastewater is reused in urban agriculture on readily accessible land, the impact in terms of GHGs is not significantly different when compared to uncontrolled release of untreated wastewater. As seen in figure 4-9, only 1% of the nitrogen is being captured in available agricultural land. This is due to results being highly sensitivity to the amount of land available. Sensitivity to a different variety of crop (paragrass, a tall grass used as animal feed) was also tested. However, the resulting emissions were not much different and are not shown here.

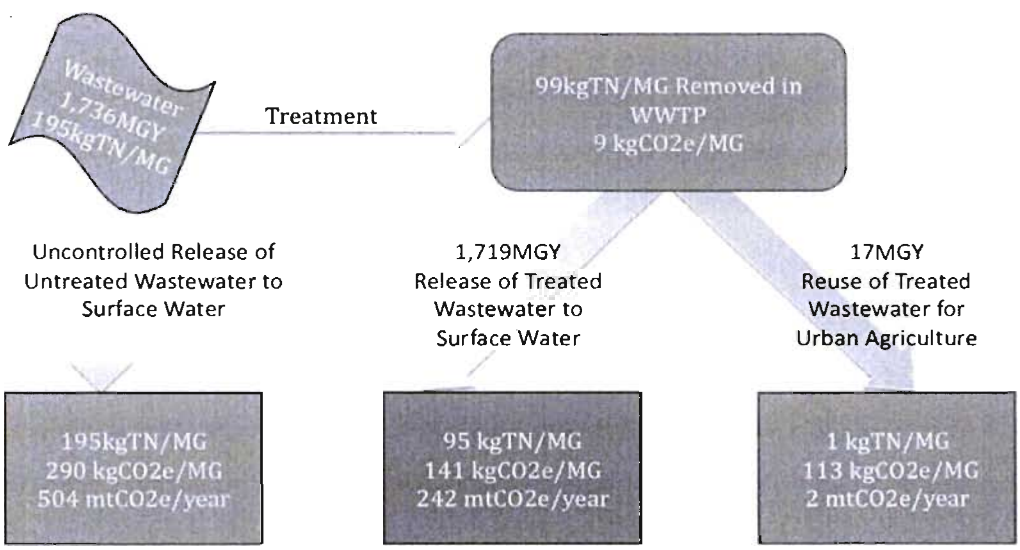


Figure 4-9: Wastewater nitrogen flows in the two core scenarios

The amount of land easily accessible (with minimal infrastructure) for treated effluent reuse in urban agriculture was estimated to be 5,525m^2 and indicated in figure 4-10. The total treated effluent released from this plant would need approximately 561,838m^2 land for total reuse in urban agriculture. There is this much land available near to NWWTP (in figure 4-10), but mixing with nearby streams would be difficult to avoid.

Figure 4-10: Land easily accessible for NWWTP treated effluent reuse in urban agriculture.

Avoided fertilizer and avoided electricity due to biogas reuse decreases the total GHG impact by 5% each. In the case that all NWWTP effluent could be directly reused in urban agriculture, the GHG impact could be decreased by about 20% (figure 4-11).

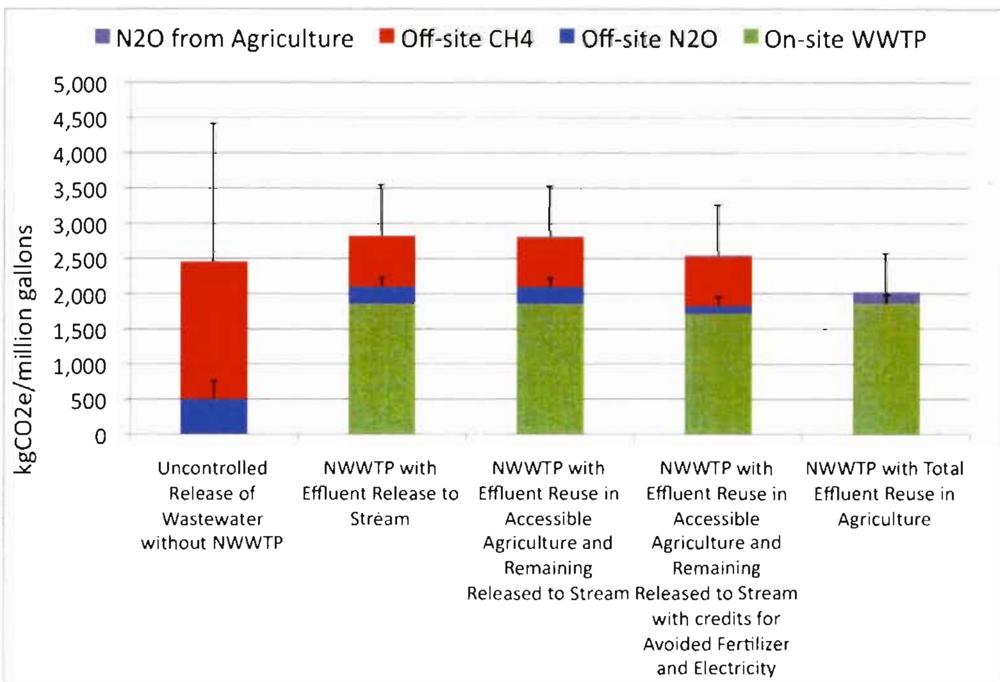


Figure 4-11: Consequential LCA results comparing the GHG emission impact from releasing untreated wastewater to various interventions

In figure 4-11, in last two cases, avoided emissions from electricity as a result of biogas reuse was subtracted from the on-site emissions. However, in the second to last case, avoided emissions due to no fertilizer use was subtracted from the off-site N_2O emissions, while in the last case, the avoided emissions were subtracted from N_2O from agriculture.

4.5 Insights and Recommendations for Future Work

The following insights were seen from the results in this chapter:

- Wastewater treatments plants are effective at removing pathogens (99%), BOD (81%), and solids (93%) from influent wastewater while retaining a high amount of nutrients and using a low amount of electricity per gallon wastewater treated.

- Energy recovery potential exists for WWTPs but is not used.

- A system-wide analysis shows that nutrient recovery from wastewater agriculture is highly dependent on the flow rate of wastewater and associated land available. Nutrient recovery was relatively small in terms of the percentage nutrients used versus the total nitrogen discharged. Dried sludge, in contrast, could be a more successful avenue for nutrient recovery as it can be distributed more safely and easily. However, a market for the sludge must first be established.

- System-wide greenhouse gas emissions with and without the use of WWTP were not very different, given the range of model uncertainty. However, energy investments did reduce BOD, COD, and pathogens.

Further exploration of many of the parameters would be useful for this study. Notably, the reasons for the unusually high influent nitrate concentration are not well understood. Measurements of influent nitrate at all four WWTPs would help to understand the variation in the sewer system. The same measurements taken before and after a storm could provide insight to the sources of the high nitrate concentrations. Also, the flows throughout the WWTP, that have been requested, would be useful for further quantifying process emissions.

In the next chapter, the fate of pathogens from wastewater reuse for urban agriculture will be discussed.

5. Measuring Water and Food Relationships: A Site Study

5.1 Introduction

As discussed in chapter 1, the life-cycle benefits and costs arising from net energy investments and net greenhouse gas (GHG) emissions in newly implemented WWTP infrastructure have not been quantified for WWTP effluent reuse in urban agriculture.

This study takes an urban agriculture perspective to evaluate water savings, nutrient delivery, and pathogen reduction achieved for irrigation of three different urban agriculture plots. The three irrigation waters of differing nutrient and pathogen qualities were sourced from:

1) Groundwater from a borewell (50 feet deep),
2) Treated effluent from the wastewater treatment plant, and
3) Untreated water in surface streams.

5.2 Site Selection and Study Design

Several sites where wastewater agriculture was used were visited within Hyderabad by a team of researchers including Miller, Ramaswami, and Amerasinghe, to assess crops being grown and to speak with farmers. Sites visited included a borewell irrigated site in Kachivani and a wastewater site in Peerzadiguda and finally, the Nallacheruvu site.

5.2.1 Site Selection

The farming site at Nallacheruvu was chosen for the following reasons: 1) its co-location of WWTP and urban agriculture, 2) ready access to three different qualities of water, 3) the availability of an experienced farmer, and 4) the HMWSSB gave permission for use of the study site and were willing to share data for NWWTP. The study took place during the dry-season in March- May

2010, when water levels were at their lowest, and wastewater was the least diluted with storm water.

Figure 5-1: Aerial view of NWWTP showing co-location of urban agriculture plots. 1: groundwater; 2: NWWTP effluent; 3: untreated surface water. Source: Google Earth, Imagery date April 5, 2010.

The **crop of interest** was spinach because it is frequently grown in this area and is a leafy green vegetable commonly eaten by people in India. Other crops grown in the region include mainly paragrass, an animal feed. The WHO irrigation guidelines are often most stringent for leaf crops because they are eaten raw in many parts of the world (World Health Organization 2006). However, it is recognized in this study that most vegetables in India are cooked and not eaten raw (Khanum, Siddalinga Swamy et al. 2000). Palak (spinach in Hindi) seeds were bought from a local agricultural store in Uppal (south-east neighborhood of Hyderabad). The farmer preferred an "All Green" variety that is known to grow well in hot temperatures. The species is actually *Beta vulgaris* and a heirloom variety of chard that is native to India (BackyardGardener.com 2010; EvergreenSeeds.com 2010; Singh and Agrawal 2010), but will herein be referred to as spinach.

The **pathogens of interest** in the crop are *Escherichia coli* (*E. coli*) and nematode ova (eggs) (Roundworm *Ascaris lumbricoides* (*Ascaris*); and Hookworm: no distinction was made between Old World, *Ancylostoma duodenale*, and New World *Necator americanus* hookworm). *E. coli* and nematode eggs are commonly used to indicate wastewater contamination and associated health risks (Cifuentes 1998; An, Yoon et al. 2007; Mara, Sleigh et al. 2007; Ensink, Blumenthal et al. 2008). Nematodes pose a high health risk when compared to other pathogens due to their infective dose being small, their ability to live longer in the environment, and the fact that humans generally have low immunity to them (Gaspard, Ambolet et al. 1997; World Health Organization 2006).

For **nutrients**, the focus was on the primary macronutrients (nitrogen, phosphorus, potassium) that are essential for plant development. Nitrogen (N) is utilized for: all proteins, enzymes, metabolic processes involved in the synthesis and transfer of energy; chlorophyll; rapid growth, increasing seed and fruit production; and improving the quality of leaf and forage crops. Phosphorus (P) is utilized for: photosynthesis; formation of all oils, sugars, starches; transformation of solar energy into chemical energy; proper plant maturation; withstanding stress; and rapid growth. Finally, potassium (K) is utilized for: building of protein, photosynthesis, fruit quality and reduction of diseases; and K levels are usually higher than others to reflect parent material (igneous rocks ~50,000ppm) (North Carolina Department of Agriculture and Consumer Services).

Furthermore, organic carbon is an important part of soil for: crop yield, soil fertility, soil moisture retention, aeration, nitrogen fixation, mineral availability, disease suppression, soil composition, and general soil structure (Leu 2007). Total organic carbon can also be measured in water and is expected in water with sewage contamination. BOD and COD were also measured in irrigation water.

pH, electrical conductivity (EC), and total suspended solids (TSS) were of interest in this study because they can affect and inform on other parameters. Levels of pH in water and soil are known to affect availability of nutrients: at low pH, macronutrients tend to be less available, while at high pH, micronutrients tend to be less available. EC, or the capacity of the media to conduct electrical current, is directly related to the amount of solids dissolved in that media (soil or water). EC can affect soil texture, cation exchange capacity, drainage conditions, organic matter level, salinity, and subsoil

Figure 5-2: Farmer, Chandriah, and translator, Aruna, at the untreated surface water plot

characteristics (Grisso, Alley et al. 2009). TSS is simply a measure of the amount of solids that are not dissolved in the water, and can relate to oxygen demand, turbidity, and organic matter in water.

5.3 Study Design

- The same crop was grown in three different sites with three widely varying source water qualities to compare the impacts of water quality on crop quality (food pathogens) and productivity.

- Plots were co-located so that the same farmer could cultivate them and the same practices could be used.

- The crops were grown over the same time period and irrigated at regular intervals by the farmer.

- The researcher observed irrigation events at least twice per week and communicated with the farmer using a translator. The farmer was instructed to treat all three plots in the same manner, and in particular, not to fertilize any one plot if it was doing poorly.

- The researcher measured flow rate at the start of each irrigation event, and sampled the water throughout irrigation events. Water from the groundwater plot was sampled near the end of the irrigation event to ensure proper purging of stagnant well water.

- Some water, soil, and crop samples were delivered to local laboratories for analysis as shown in gray below. Other samples were analyzed by the researcher at lab facilities provided by the International Crop Research Institute for the Semi-Arid Tropics (ICRISAT) with a cooperative agreement with UC Denver.

- The researcher tested for pH, electrical conductivity, and total suspended solids in water, and *E. coli,* total coliforms, and *Ascaris* and Hookworm ova in water, soil, and crop.

- In addition, the researcher had purchased Hach kits in the US to complete nitrogen, phosphorus, and potassium testing. But because of customs regulations banning transport of chemical reagents, these tests had to be outsourced to a lab in India.

Table 5-1: Tests done over one crop growing cycle. Gray: tests outsourced to a lab; white: tests done by this researcher. All sampling, transport, and sample preparation was also done by this researcher.

	Groundwater	Plot Treated Effluent	Untreated Surface
Pre-analysis	**Tests done**		
Soil Physical Characteristics	X	x	x
Soil Nutrients	X	x	x
Irrigation Water Pathogens	X	x	x
Irrigation Water Nutrients	X	x	x
Dynamic Monitoring	**Frequency of tests over study**		
Water Quantity (volume)	9*	9*	9*
Water Pathogens	9*	9*	9*
Water Nutrients	9*	9*	9*
Soil Pathogen (*E. coli*/Nematode)	mid, end	mid, end	mid, end
Soil Nutrients	3	3	3
Soil Water Nutrients	mid	mid	mid
Crop Quantity (weight)	End	end	end
Crop Pathogen (*E. coli*/Nematode)	mid, end	mid, end	mid, end
Crop Nutrients	mid, end	mid, end	mid, end

* Farmer irrigated about every two days, depending on weather. Water quantity, pathogens, and nutrients were recorded by the researcher every alternate irrigation event

5.3.1 Composites and Replicates

- The researcher gathered composite water samples during the observed irrigation events (over 5-15 minutes depending on water flow rate) for water quality analysis, and crop and soil samples before irrigation events.

- Crops were sampled at midpoint and endpoint (n=3 from each plot). The endpoint represents harvest conditions.

- Duplicates were run by the researcher for *E.coli* at every tenth sample, regardless of the media, throughout the study as a method of doing quality assessment/quality control (QA/QC). Control plates (run with dilution water) were prepared along with every set of samples in order to ensure that sterile techniques were used.

- Independent laboratories did their own QA/QC with blanks and standards.

- When weighing crop bundles at harvest, a composite sample of 20 bundles was taken to the lab and weighed. Then, an average weight per bundle could be determined.

- Three grab samples (100g spinach each) from the larger composite sample from each plot were taken for *E.coli* testing, then composited again for nematode testing.

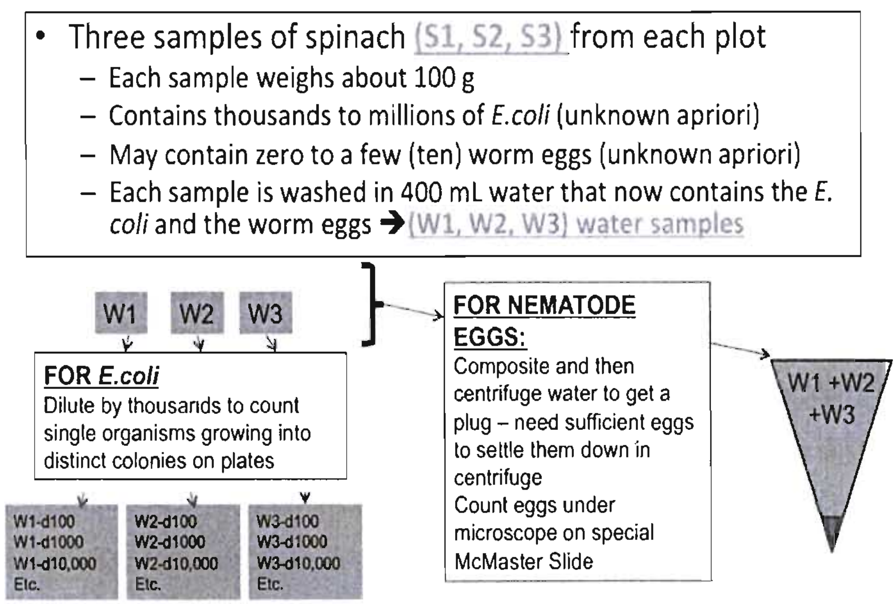


Figure 5-3: Crop pathogen measurement overview: challenge of dilution versus concentration

Next, the methodologies to carry out this study will be discussed.

5.4 Site Study Methodology

Pre-analyses of water and soil quality were done to determine the most-suitable plots for this study. The purpose was to have certain factors for the three plots to be as similar as possible, such as soil characteristics, location, and farmer practices, thus isolating the impact of water quality on urban agriculture. Plots were chosen according to pre-analysis results. After proper preparation of the plots, including wetting, weeding, and plowing, the seeds were planted and routine testing was done throughout one growing cycle. Pre-analysis is described first, then dynamic measurements.

5.4.1 Site Preparation and Pre-Analyses

Before choosing the exact location of the plot, water and soil parameters were tested. Water tests were taken from many locations to get initial measurements of nutrients and pathogens in the area. Soil samples were taken from each potential plot location and tested for physical characteristics and nutrient content.

For analysis of soil physical characteristics, samples were taken from four random spots within each potential plot with the appropriate soil core tools throughout from 0-15cm and 15-30cm, then combined to form a composite sample for each soil layer. Soil texture (distribution of particle sizes), bulk density, wilting point and field capacity, porosity, and particle density were measured at ICRISAT using sieve analysis, moisture retention at pressures of 0.33 and 15 bar, bulk weight per volume, and particle size tests and calculations. Soil type for the three plots were similar as determined by the soil texture pyramid (figure 5-4) (National Resources Conservation Service 2011).

The plot locations were chosen so that the soil textures were similar and the three plots were co-located with their individual source of water. Their orientation to stream bed, delivery of water, direction of water flow, size, slope, and elevation were taken into consideration and were made as similar as possible. All plots were oriented with their longest side parallel to the closest stream bed (approximately 10m away for each) and every plot was irrigated by opening a channel and allowing the plot to flood lengthwise. Each plot was 12 m^2, but the untreated surface water plot was 6m by 2m while the groundwater and treated plots were 4m by 3m, due to space constraints and an effort to maintain as shallow a slope as possible. The elevation for each plot was roughly the same at 470±1m above sea level. Plots were leveled but had a slight gradient to facilitate gravity-flow irrigation.

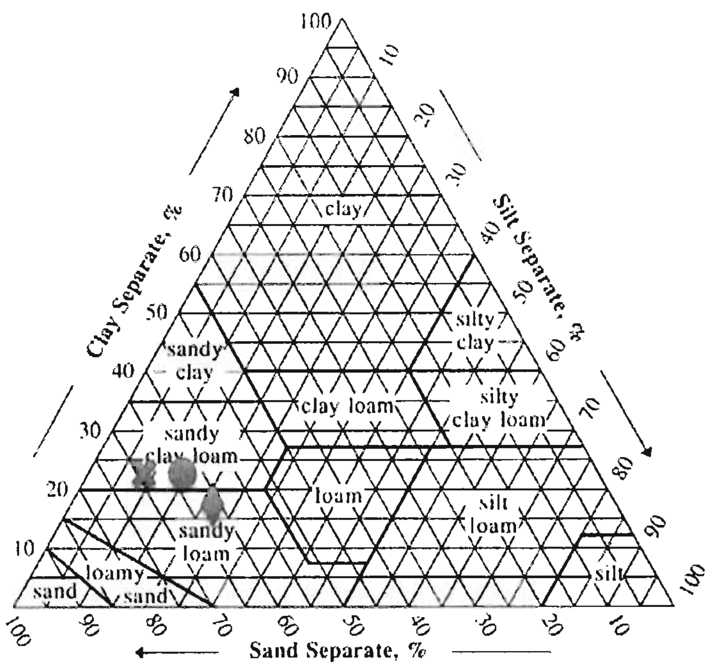


Figure 5-4: Soil texture pyramid. Groundwater plot: x; treated effluent plot: o; untreated surface water plot: ◊.

Small channels were dug to deliver water to the groundwater and treated effluent water plots, while the untreated surface water plot had an established network of channels that diverted water from the nearby surface stream. The groundwater and treated effluent water sites were wetted with the appropriate water and newly plowed 10 days before seeding. The untreated surface water plot had been cultivated with spinach approximately 1 month earlier and laid without a crop for that time.

For this study, there was much discussion on whether to amend the soils with lower nutrient content, to make them all similar. It was decided that the soils would not be amended, as the farmer, Chandriah, was confident that the crops would grow in all plots. The farmer prepared soils as per his normal practice with the appropriate water for site and plowing. He did not use additional fertilizers throughout the study.

5.4.2 Dynamic Measurements

The following parameters were measured periodically throughout one crop growth cycle from March 26- April 28, 2010:

- Irrigation water quantity
- Irrigation water quality
  - pH, EC, TSS
  - BOD, COD, TOC
  - Nutrients: Nitrogen, phosphorus, and potassium
  - Pathogens
    - *E. coli* and Total coliform
    - Nematode ova (*Ascaris* and Hookworm)
- Soil quality
  - pH, EC, TOC
  - Nutrients: Available and total nitrogen, phosphorus, and potassium
  - Soil water nutrients: Nitrogen, phosphorus, and potassium
  - Pathogens (same as in water)
- Crop quantity
- Crop quality
  - Nutrients: Nitrogen, phosphorus, and potassium
  - Pathogens (same as in water)

The methods for each of these tests are described next.

5.4.2.1 Water Quantity

In order to standardize the volumes of water over all three plots during flood irrigation (by both the researcher and the farmer), various methods were attempted. Because the groundwater and treated effluent water plots had water being delivered through a pipe, which flowed into a channel, a time/volume method was used. In this method, the amount time that it took to fill a known volume ("bucket") was noted at least twice a week at time of irrigation. The flows were not too fast and this method worked well.

For the untreated surface water plot, the water was delivered in a channel. To measure volume here, a "float" (a cap from a water bottle) was used and the time that it needed to travel a known distance was measured. Then the flow was calculated by multiplying velocity by cross-sectional area of the channel. Each time this method was used, the banks and the bottom of the channel were cleaned from vegetation to ensure that the initial cross-sectional area measurements were retained. This method was always done three times and the average was taken.

Figure 5-5: Preparation of the channel, with help of the farmer, Chandriah, leading to the untreated surface water plot

For the groundwater and treated effluent water plots, the "bucket" method was compared to the "float" method within the channel leading the plot. With these rough calculations, the float method was found to be, on average, within

9% (standard deviation of ± 8%) by volume of the bucket method for measuring water volume.

5.4.2.2 Water Quality

Sampling of water was done in the same way throughout the study. Any materials for *E. coli* detection were sterilized prior to sampling or testing. Sterile procedure and techniques were strictly followed for all *E. coli* tests. All other sampling vessels and materials were always cleaned with soap and water and well rinsed with distilled water prior to sampling or testing. Water samples were always taken at points where the water was well mixed. This was determined visually by fast moving, turbulent flow. Samples were taken from below the surface of the water at 40-60% of the water depth to minimize settling of solids. Care was also taken not to disturb sediment before or during sampling. Sampling vessels were always filled to the top so as to minimize air space. All samples for *E. coli* testing were kept on ice and the tests were done the same day, as soon as possible. Samples for nutrient testing were kept on ice, transferred to a refrigerator in the appropriate lab, and the test was done as soon as possible.

Generally, irrigation water may not be considered as a significant source of nutrients or pathogens. However, because the irrigation water in this study contains wastewater, it is an important part of the agricultural system.

The following tests were done on water:

pH, EC, TSS: Measurements of pH and electrical conductivity were done by this researcher using a Oakton® Multi-Parameter Meter. The instrument was periodically calibrated for temperature and pH standards. TSS was measured by filtering water through a filter paper and then drying it at 3-4 hours at 65°C (the only oven available) (Standard Method 2540 D).

TOC, BOD, and COD: Total organic carbon was tested to determine the amount of organic matter in the irrigation water. The test was carried out on a Total Organic Carbon Analyser TOC-V_{CPN} (manufactured by Shimadzu Corporation). BOD_5 (Standard Method 5210 D) and COD (Standard Method 5220 B Open Reflux Method) were also done 2 and 3 times, respectively, for irrigation water. All these tests were contracted to the Environmental Protection Training and Research Institute (EPTRI) labs in Hyderabad.

Nutrients NPK: Water was tested for plant available dissolved inorganic nutrients such as: inorganic soluble nitrogen (NH_4^+-N Standard Method 4500-NH_3 C and NO_3^--N Standard Method 4500-NO_3^- E), soluble phosphorus (Standard Method 4500-P D), and soluble potassium (Standard Method 3500-K B). All tests were carried out using Standard Methods for Examination of Water and Wastewater (American Public Health Association, American Water Works Association et al. 2006) at ICRISAT. Nitrite (Standard Method 4500-NO_2^- B) in all three irrigation waters was tested once to confirm that nitrite concentrations were low and could subsequently be ignored as insignificant.

Pathogens: Hach's membrane filtration method for *E. coli* and total coliform detection was the most accessible and precise method for fecal indication. This method involved diluting the water sample by factors of 10^1, 10^2, 10^3, 10^4, 10^5, 10^6, until each pathogen could be assumed to develop into separately visible colonies over an incubation period of 20-24 hours. The filter paper on which the pathogens are filtered is placed into a sterile petri dish with nutrient both that stains E. coli red/purple and total coliforms blue. Then, the dilution factors can be applied to scale back to the original concentration of these bacteria in source water. See Appendix B for method details.

For detection of *Ascaris* and hookworm eggs, a method developed by IWMI (which was adapted from Ayers and Myer 1996) was used. This involved gathering large samples, at least 5 liters of each irrigation water and leaving them to settle overnight in the lab. The next day, a multi-step centrifugation protocol was done to separate sediment and eggs from the water. Then, the sediment containing the eggs was suspended in zinc sulfate, and a McMaster slide was used to identify and count the eggs under a microscope (Ayres and Mara 1996). A more detailed description of these methods can be found in Appendix B.

5.4.2.3 Soil Quality

Soil samples were taken from random spots within the plot (first carefully removing the top 3cm and extracting the sample). For chemical and physical testing, soil was extracted with a soil core tool. For E. coli testing, soil was extracted with a small shovel. Soil samples for *E. coli* testing were kept on ice and the tests were done the same day, as soon as possible.

pH, EC, TOC: Total organic carbon was also done to determine the amount of organic matter in the soils. Soil pH and EC were also done by ICRISAT. Methods for all of these tests can be found in Soil Science Society of America and American Society of Agronomy 1996.

Nutrients: Available and Total NPK: The soil quality parameters tested were for both plant available and total nutrients such as: mineral (NH_4^+-N and NO_3^--N) and total nitrogen, Olsen and total phosphorus, and exchangeable and total potassium (Soil Science Society of America and American Society of Agronomy 1996). ICRISAT carried out these test and included QA/QC with a known standard sample from the International soil analytical exchange. The results from this standard sample were always found to be within the expected range for nitrogen, phosphorus, and potassium. Also, duplicates run were always found to be within 99% of each other.

Lysimeter Soil Water and NPK: Soil water was collected with two lysimeters per plot, randomly placed near the inlet of water to the plot and another randomly placed farthest from the inlet to plot (Soilmoisture Equipment Corp. 1900L12-B02M2). These soil water samplers were installed according to the method provided by Soilmoisture and placed under a vacuum for over a period of 8 hours in each plot. The samples were extracted and taken to the EPTRI lab for total nitrogen (KEL Plus Classic DX TN Analyzer), total phosphorus (Standard Method 4500-P D), and total potassium (Standard Method 3500-K B) tests.


Figure 5-6: Lysimeters installed in groundwater plot.

Pathogens: *E. coli* testing was done by slightly modifying of Hach's membrane filtration method. Using sterilized materials, 10 g of soil was added to 95mL of 0.8% NaCl, covered with parafilm, and placed on a shaker for 60 minutes. Under a sterile vacuum hood, this liquid was then filtered through a strainer to remove large particles that would clog the membrane. With this solution, appropriate dilutions were done and the regular method was followed as described in section 5.4.2.2.

The method for nematode ova identification in soil was adapted from Zenner et al. for this study (Zenner, Gounel et al. 2002). 50g soil was added to 200mL distilled H_2O. This was mixed thoroughly with a metal rod for 20 seconds and filtered through a coarse sieve. The solution was divided into tubes and centrifuged. Then, sediment from one tube was resuspended in the floation solution, magnesium sulfate, and the top of the tube was covered with a glass cover slip. During a second centrifugation step, the eggs floated to the top of the tube and stuck to the coverslip. Eggs on the coverslip could be counted under a microscope. This second centrifugation step was repeated until eggs were no longer found.

5.4.3 Farming, Irrigation, and Harvesting Practices During Study

After the plots were chosen, the source of irrigation water was the major difference in practice between the plots. The farmer, Chandriah, cultivated all three plots and his normal practices were followed for plowing, seeding, irrigation, pest control, and harvesting. Plowing was done by hand with a hoe. Seeds were broadcasted by sweeping motions over the plot, then raked into the soil, then flooded with water. The plots were irrigated about every 2-4 days, as was deemed appropriate by the farmer, for weather and soil conditions.

On the 10th day of the study (about 1/3 into the study) a worm was attacking and causing a lot of damage to the treated effluent plot and the untreated surface water plot. An insecticide (phorate: http://www.hyderabadchemicals.com/hyfort.html) (US EPA 2010) was used on this day, once only, to kill this insect and the farmer was sure that the crop would be lost if this action was not taken. This action may have affected the nematode population in the soil, but it is unknown whether it affected nematode ova. Because the farmer uses this chemical regularly when there are insect problems, it is likely that it does not kill nematode ova, because there are many in the soil. Soil samples were taken for *E. coli* one week later, for nutrients one and a half weeks later, and for nematode ova two weeks later. This treatment will not affect the water samples as the water is taken at the entrance to the plot, before flowing across the plot surface.

At time of harvest, the leaves were cut at the base (before the roots) and gathered into bundles. To hold them together, they were wrapped with a string. Then, the bundles were piled on the ground, under a cloth that had been wetted (by any nearby water) to prevent wilting.

5.4.3.1 Crop Quantity

The average height of spinach plants in each plot was determined by taking several measurements in the plot twice per week. At time of harvest, the amount of harvested bundles were counted and many bundles were weighed to determine an average weight per bundle. Final harvested weights and approximate gC/m2 were found by using 42.5g carbon/kg plant to find a percent C by weight for net production by gC/m^2 (National Council on Radiation Protection and Measurements 1983; Noguchi and Terashima 2006).

5.4.3.2 Crop Quality

Spinach growing in each plot was only sampled 2-3 times for each test throughout the study in order to minimize effects on growth. Any stepping within the plot and harvesting of plants affected the overall production.

Figure 5-7: Harvesting spinach, which is bundled and placed under a cloth wetted with nearby water, usually untreated surface wastewater, to prevent wilting.

Nutrients NPK: The following tests were carried out for the plant tissue from each plot twice over the course of the study: total nitrogen, total phosphorus, and total potassium. Prior to analysis, this researcher separated the leaves and roots dried them at 65°C until they could be ground by mortar and pestle into a powder. Then, this powder was taken to the ICRISAT lab for testing.

Pathogens: While spinach was being harvested by the farmer, this researcher used hand sanitizer and used sterile procedures to harvest a few random samples. *E. coli* tests were done for both sets of samples (sterile-harvest and farmer-harvest). The farmer harvest represents the conditions in the field and what would be sold at market.

The method was customized by IWMI for identifying *E. coli* and nematode eggs on the surface of crops. 100g of spinach were shaken vigorously with 400mL of phosphate buffered saline solution in sterile bags. Then, the solution was poured into sterile beakers and the methods for *E. coli* and nematode eggs detection in water were followed (as described in section 5.4.2.2 and Appendix B).

5.4.4 Sustainability Assessment Methodology

Based on the site study results, five sustainability assessments were made as described in this section. The tradeoffs between them are displayed in the sustainability diamond. This approach has been used by others (Pearce and Vanegas 2002; Aubin, Papatryphon et al. 2009).

5.4.4.1 Energy and GHG

Methodologies for estimating these impacts for the treated effluent and untreated surface water plot can be found in Chapter 4. For the groundwater site, the energy use and related GHG was only for pumping water from a well. This was estimated by calculating the work required to lift the volume of water used for the groundwater plot (equation 5-1).

$$W = (d_{H_2O} * V_{H_2O}) * g * h * \varepsilon_{pump} \qquad \text{(Equation 5-1)}$$

where: W= work (J or kg*m^2/s^2)

d_{H2O}= density of water: 1kg/L

V_{H2O}= volume of water lifted, 6,975 L over one month (one growing cycle)

g= acceleration of gravity: 9.81m/s^2

h= height water lifted: 15.24m

ε_{pump}= efficiency of the water pump: 21% (Saravanan 2010).

The pump efficiency used was for agricultural pumps in India and ranged from 20-22% and the average was used (Saravanan 2010). The overall energy use for groundwater pumping was over 10-one month growing cycles for one year, yielding electricity use per year.

5.4.4.2 Pathogen Health Risk Assessment

When crops have matured and farmers harvest them to send to urban markets, various practices can affect the final contamination levels of vegetables leading up to consumption. These are:

- Die-away due to chemical, physical, and biological factors: UV irradiation from sunlight, desiccation, adsorption, settling, and biological competition (Fattal, Lampert et al. 2004).
- Amount of water sticking to external surface of vegetable: can vary widely as this amount is 0.36 ml/100 g for cucumbers and 10.8 ml/100 g for long-leaf lettuce (Shuval, Lampert et al. 1997).
- Increasing the amount of days between the last irrigation and consumption can cause rapid pathogen die-off. However, this technique also decreases freshness, therefore possibly decreasing market value (Blumenthal, Mara et al. 2000).
- Post-harvest re-contamination by contact with contaminated soil, other produce, water (washing or refreshing/splashing water in markets), or via contaminated surfaces (Hamilton, Stagnitti et al. 2006; Andoh, Abaidoo et al. 2009).
- Kitchen practices: washing, peeling, and cooking (Hamilton, Stagnitti et al. 2006).

To determine the health risk of exposure to pathogens when consuming vegetables irrigated with treated and untreated wastewater, the basic model for ingesting pathogens through drinking water was used as a first step (Haas, Rose et al. 1993; Sakaji and Funamizu 1998). Haas et al. (1993) developed a basic model (using Equations 5-2 and 5-3) for approximating the risk of infection and disease from ingesting pathogens through drinking water:

$$P_I=1-[1+N/N_{50}(2^{1/\alpha}-1)]^{-\alpha} \qquad \text{(Equation 5-2)}$$

where: P_I= the probability of infection by ingesting pathogens in drinking water;
N= number of pathogens ingested: 100g per week for one year;
N_{50}= number of pathogens that will infect 50% of the exposed population due to the same event: $8.6*10^7$ for *E. coli* (Haas, Rose et al. 1999) and 859 for *Ascaris* (Navarro, Jiménez et al. 2009);
α= slope parameter, or infectivity constant (See Table 5-2 and Figure 5-8): 0.1705 for *E. coli* (Shuval, Lampert et al. 1997; Asano, Burton et al. 2007) and 0.104 for *Ascaris* (Mara and Sleigh 2010).

Because infection does not always cause disease, the probability of contracting a disease is estimated by:

$$P_D= P_{D:I}*P_I \qquad \text{(Equation 5-3)}$$

where: P_D= the probability of an infected person becoming diseased or ill;
$P_{D:I}$= the probability of an infected person developing clinical disease: 0.5 used for *E. coli* (Haas, Rose et al. 1999) and 1 used for nematode eggs (Mara and Sleigh 2010).

Equations 5-2 and 5-3 were used directly to calculate health risk from ingesting 100g spinach per week for one year. In this study, the amount of *E. coli* and nematode eggs on the surface of spinach were quantified. Because microorganisms were found to remain on the surface of vegetables and were not taken up into the actual tissue (Shuval, Lampert et al. 1997), this method was accurate for estimating pathogen exposure when vegetables are eaten raw.

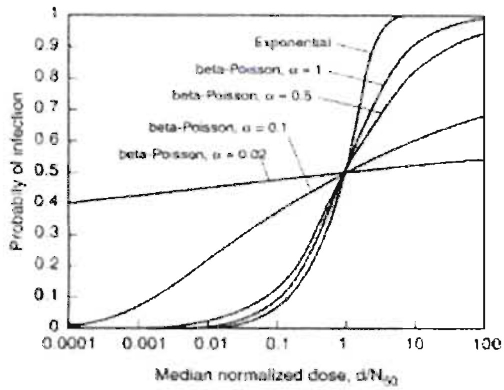


Figure 5-8: Comparison of beta-Poisson and exponential dose-response functions. d= mean ingested dose. Source: Adapted by Asano et al. 2007 from Haas and Eisenberg 2001.

Table 5-2: Summary of dose-response slope parameter for various enteric pathogen ingestion studies. Source: Adapted by Asano et al. 2007 from Regli et al. 1991, Ward et al. 1986 and Black et al. 1988.

Constituent	α
Virus	
Echovirus 12	0.374
Rotavirus	0.253
Poliovirus 1	0.1097
Poliovirus 3	0.409
Bacteria	
Salmonella	0.33
Shigella flexneri	0.2
Escherichia coli	0.1705
Campylobacter jejuni	0.145
Vibrio cholerae	0.097

This method also accounts for any die-off or recontamination occurring pre-harvest as it is an actual measurement rather than a calculated risk. The probability of disease is calculated for eating 100g of spinach per week over a one year time period. To estimate the amount of spinach or palak that a person living in India may eat per year, the Food and Agricultural Organization's (FAO) food balance sheet for India was used (FAOSTAT 2000). Spinach was estimated to make up about 10% of the "other vegetables" category (this category is separate from tomatoes and onions) in a person's diet in India. This roughly equates to a person eating 100g (about two bunches) of spinach per week over one year. This consumption was used to estimate risk to an individual's health per year.

5.4.4.3 Productivity

To determine the overall productivity of the plots, the total harvested weight of marketable crop from each plot was used. As each plot was the same area ($12m^2$), a weight per area result could be calculated.

5.4.4.4 Water Savings

Because soils have great influence on water use in flood irrigation, the volumes of water used in this study were likely to be higher than the amount of water actually needed by the crop to grow. The crop water requirement was a better tool to determine the amount of water savings. The crop water requirements for spinach grown in China, Mexico, Thailand, and Indonesia were averaged (FAO 2011). This resulted in 94 gallons/m^2 irrigation water needed for spinach growth and this estimate was used for India. The quantity of crop produced per m^2 was used to determine water use for 10 growing cycles in one year.

For the groundwater plot, there is assumed to be zero water savings. The treated effluent and untreated surface water plots were assumed to be equivalent to the amount of irrigation water on these sites that would have otherwise come from groundwater.

5.4.4.5 Cost of Infrastructure

The cost of building NWWTP was almost $2,912,785 (2007 USD) (12 crore Indian rupees) in 2007 and was amortized over 30 years. The approximate cost to drill a well, pour a concrete base, and buy and install a pump mechanism in Africa was $6,000 USD (The Water Project 2011). This was used to estimate the costs for the groundwater well in India. A useful lifetime of 10 years was assumed. Neither of these costs include operating expenses. Untreated surface water does not have infrastructure associated with it, therefore its cost is negligible.

5.5 Results

5.5.1 Pre-Analysis Results

5.5.1.1 Water and Soil Quality

There were large differences between the three types of water for almost every parameter, while the differences between the parameters for soil were more similar. As seen in figure 5-9A, pathogen levels in the three different source waters ranged from around 500 colony forming units (CFU) per liter (L) to almost 10,000,000 CFU/L. Nitrogen, phosphorus, potassium (NPK) levels are shown in figure 5-9B.

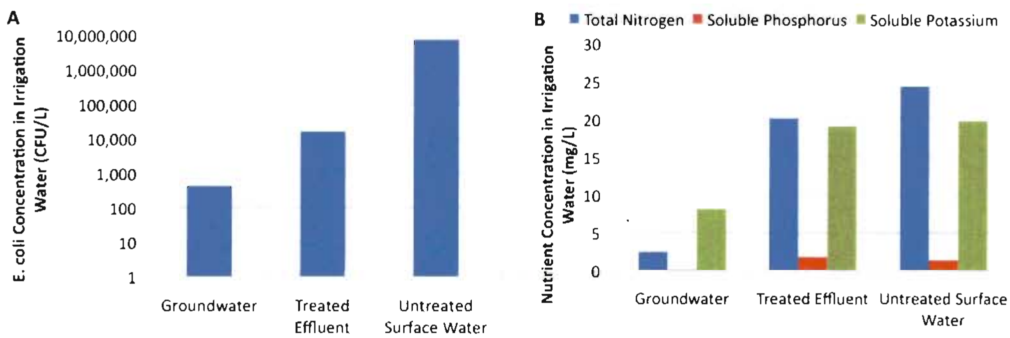


Figure 5-9: Initial measurements of *E. coli* (A) and NPK nutrients (B) in irrigation water

Dynamic measurements to relate differences in water quality in the three different sites to crop quality are described next.

5.5.2 Dynamic Measurements

5.5.2.1 Water Quantity and Quality

The irrigation water quantity varied between the three plots for various reasons. The groundwater plot was undisturbed and perhaps, further compacted, during construction of WWTP. Higher bulk density contributed to slow infiltration and water saturated the soil quickly. The treated effluent plot was greatly disturbed during WWTP construction. Consequently, for the first few irrigations, both before and for a couple events after seeding, water sank into ground (possibly into a hole deep under plot). Early in the study, horizontal water movement was slow across this plot, but as soil became more compacted, the infiltration rate decreased due to the higher soil bulk density. The untreated surface water plot was not part of WWTP construction and was cultivated by a farmer continuously for many years. Lower bulk density, higher moisture retention and water movement caused faster infiltration into soil. See table 5-3 for total, average, and standard deviation of irrigation volume for each plot.

Table 5-3: Volumes of irrigation water resulting from flood irrigation of the different plots

Plot	Average volume ± standard deviation applied at each irrigation event (gallons)	Total Loading (gallons)
Groundwater	123 ± 21	1,843
Treated Effluent	407 ± 260	5,295
Untreated Surface	345 ± 193	4,144

5.5.2.1.1 Water Quality

The water quality, in terms of **pathogens** over the 4½ week irrigation period, varied for each of the plots, but was still significantly different across the three plots. There were at least 2 orders of magnitude differences between the *E.coli* content between the three irrigation waters (figure 5-10A). *Ascaris*, the most commonly found of the nematode ova, were in higher and widely varying concentrations in the untreated surface water, while the groundwater and treated effluent water had very little (figure 5-10B).

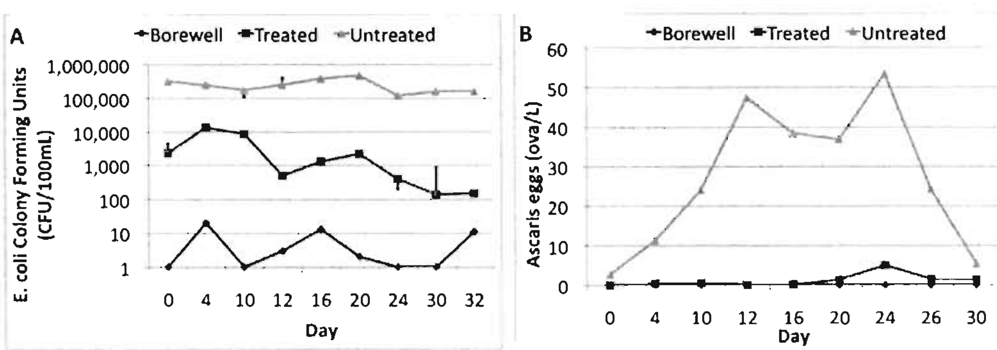
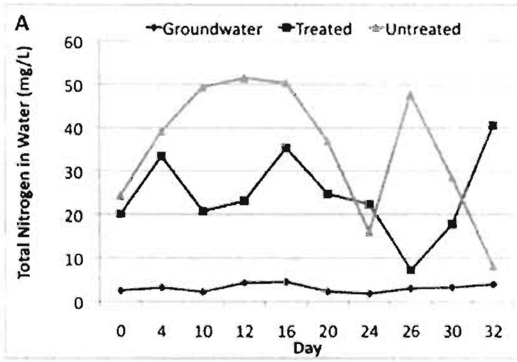
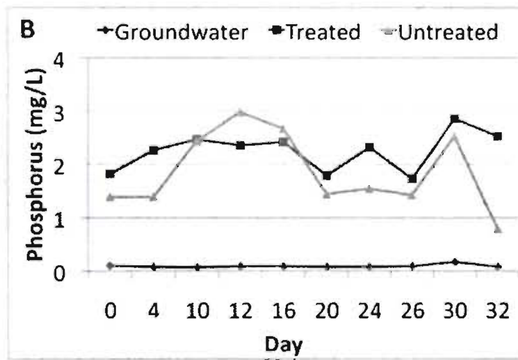


Figure 5-10: *E. coli* (A) and *Ascaris* (B) in irrigation water over the one month growing cycle

Nutrient differences between the irrigation waters were also quite different. Higher levels of total nitrogen (inorganic nitrogen plus organic nitrogen), soluble phosphorus, and soluble potassium were seen in the treated effluent water and the untreated surface water. The groundwater plot did not have as much plant available nutrients delivered at each irrigation event (figure 5-11 A, B and C).





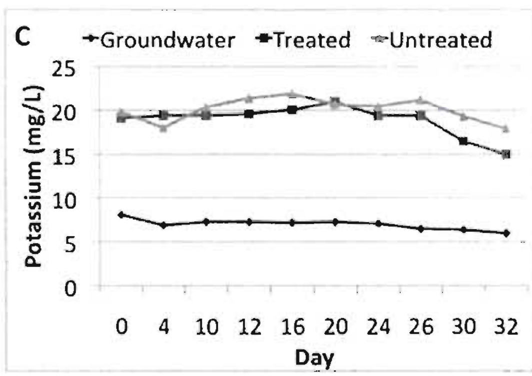


Figure 5-11: Total nitrogen (A), soluble phosphorus (B), and soluble potassium (C) in irrigation water for the three plots over one growing cycle

Combining the irrigation volume with these results, the applied loads over the entire study were determined for pathogens and nutrients. Table 5-4 shows that the total loading for nutrients was much higher in the treated effluent and untreated surface water plots, when compared to the groundwater plot. This lack of delivered soluble nutrients is thought to have limited the plants in the groundwater plot. Also, the amounts of pathogens in groundwater versus treated effluent versus untreated surface water are quite large.

Table 5-4: Total loading of nutrients and pathogens over one growing cycle (this study)

Plot	Total Nitrogen	Soluble Phosphorus	Soluble Potassium	*E. coli*	*Ascaris*	Hook-worm
	g/growing cycle			CFU/ growing cycle	ova/growing cycle	
Groundwater	324	9	744	395	0	0
Treated Effluent	5,243	567	5,071	658,994	23	16
Untreated Surface	5,685	306	3,489	40,318,444	426	61

5.5.2.2 Soil Quality (and Soil Water)

The *E. coli* content of soil was also monitored over the 4-week period and the *E. coli* content was much closer than it was in the irrigation waters. The *E. coli* content in the soil of the groundwater and treated effluent water plots were very similar throughout. The untreated surface water plot had a little more *E. coli* in the soil, but at the end of the study, they were all very similar (figure 5-12A). *Ascaris* ova were found in the soil at the midpoint and endpoint of the study, and increased slightly during this time (figure 5-12B).

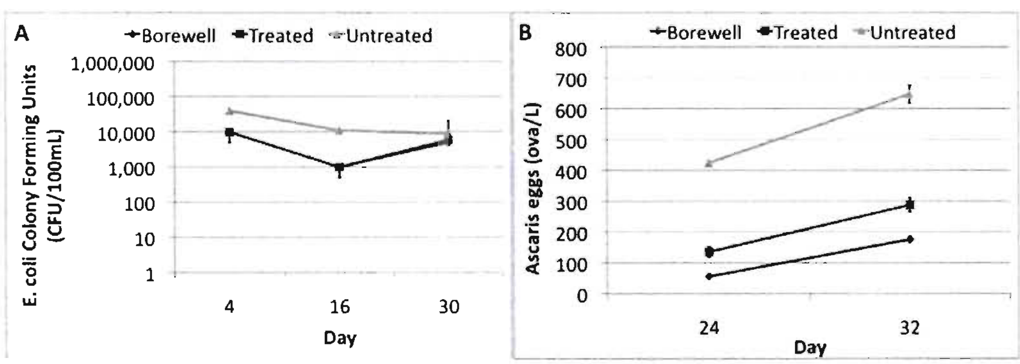


Figure 5-12: *E. coli* (A) and *Ascaris* (B) content of soil throughout one growing cycle (this study)

The macronutrient content of soil was important in determining the soil quality. All three plots were found to be within the expected range for cultivated soils of 600-5,000 mg total nitrogen/kg soil (Soil Science Society of

America and American Society of Agronomy 1996), although the groundwater and treated effluent water sites were near the lower end and the untreated surface water plot was near the high end. Here, the groundwater plot is the lowest and untreated plot is the highest, but all are within the expected range. Of the respective totals for each plot, the percentage of total nitrogen that was available to plants (mineral nitrogen), on average, was 6.7% for the groundwater plot, 3.8% for the treated effluent water plot, and 2.3% for the untreated surface water plot.

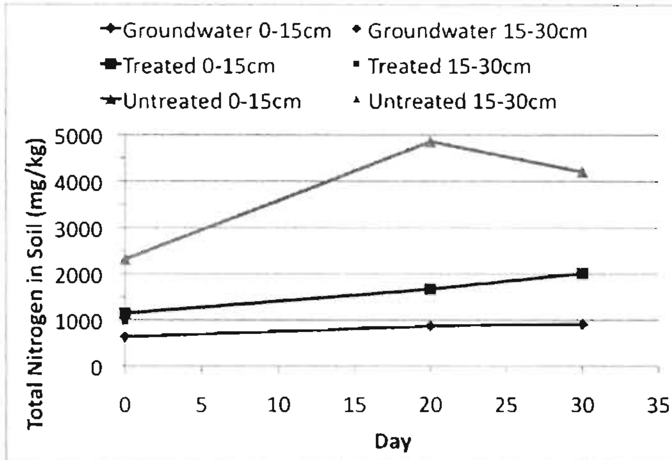


Figure 5-13: Total nitrogen in soil over one growing cycle. Soil sampling depths are noted.

For phosphorus, all plots also had levels in the soil that were within the expected range of 200-5,000 mg total phosphorus/kg soil (Soil Science Society of America and American Society of Agronomy 1996). The groundwater and treated effluent water plots had levels that were about average (600 mg total phosphorus/kg), while the untreated plot had higher levels. Of their

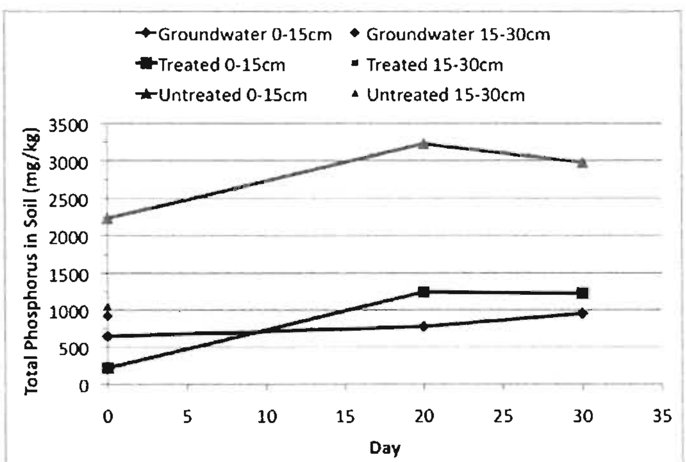


Figure 5-14: Total phosphorus in soil over one growing cycle. Soil sampling depths are noted.

respective totals, the percentage of total phosphorus that was available (Olsen phosphorus), on average, was 8.7% for the groundwater plot, 7.9% for the treated effluent water plot, and 8.5% for the untreated surface water plot.

Soil potassium was towards the high end of the expected range of 400-30,000 mg K/kg soil for all plots (Soil Science Society of America and American Society of Agronomy 1996). On average, the percentage of total potassium that was available (exchangeable potassium) were: groundwater irrigated plot: 0.6%, treated effluent irrigated plot: 0.7% and untreated surface water irrigated plot: 1.1%.

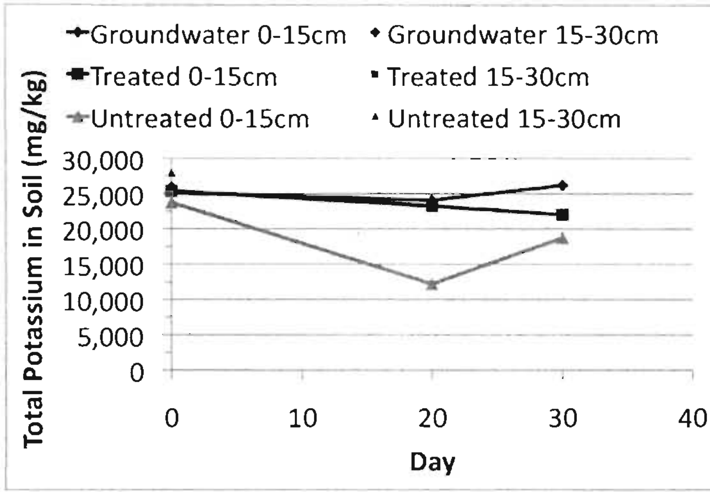


Figure 5-15: Total potassium in soil over one growing cycle. Soil sampling depths are noted.

All soils were weakly alkaline, so macronutrients are not restricted by pH.

There is not evidence that soil in any one plot was significantly nutrient limiting. However, an upward trend in the amount of nitrogen and phosphorus in each plot shows that irrigation water did contribute these nutrients to the soil. A slight decline in potassium is likely due to plant uptake.

5.5.2.2.1 Soil Water Quality

As seen in table 5-5, soil water nutrient content was not widely different. The soil water of the groundwater plot had a little less nitrogen and more potassium. This result can be expected from the soil nutrient contents for the plots.

Table 5-5: Soil water nutrient content from each plot

Plot	Total Nitrogen	Total Phosphorus ppm (mg/L)	Total Potassium
Groundwater	96	0.1	36
Treated Effluent	158	0.2	26
Untreated Surface Water	152	0.3	18

5.5.2.3 Crop Quantity and Quality

Crop quantity was determined at time of harvest in terms of crop weight (wet weight). The crop growth was visibly much less in the groundwater plot when compared to the other two plots (figure 5-16).

Groundwater Treated Effluent Untreated Surface Water

Figure 5-16: The plots at time of harvest.

The amount of sellable bundles that came from the groundwater plot was much less than the other two plots at 67 bundles (3 kg wet weight), while the treated effluent and untreated surface water plots yielded 530 (23 kg wet weight) and 573 (28 kg wet weight) bundles, respectively.

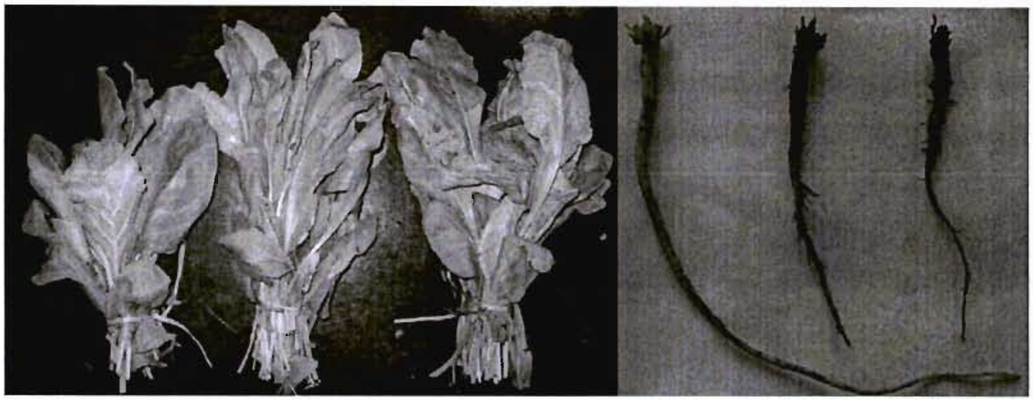


Figure 5-17: Harvested bundles of spinach and their roots. The plots from left to right: groundwater, treated effluent, and untreated surface water

While many plants in the groundwater plot were very small, some sellable bundles were produced (figure 5-17). The roots of the plants in the groundwater plot were much longer and thicker than the roots of plants in the other two plots, and suggested that the plants were seeking out water and nutrients.

5.5.2.3.1 Crop Quality

The crop nutrient content was measured on a dry weight basis. The plants in the groundwater plot had a lower nitrogen and potassium content than those in the other plots.

Table 5-6: Crop nutrient content of leaf and root

Plot	Sample	Total Nitrogen	Total Phosphorus Midpoint/Endpoint ppm (mg/kg)	Total Potassium
Groundwater	Leaf	36,800/37,500	5,400/5,500	18,100/21,300
	Root	16,700/13,700	4,100/4,000	18,400/13,600
Treated Effluent	Leaf	44,200/46,100	4,300/4,900	31,000/36,500
	Root	26,600/18,500	4,000/3,700	32,600/14,800
Untreated Surface Water	Leaf	48,800/49,000	6,200/5,500	31,500/23,700
	Root	29,200/27,800	6,200/6,800	29,900/22,800

After farmer harvest, *E. coli* on the crop did not vary much between plots and did not change radically over time (figure 5-18A). *Ascaris* ova on the crop increased in each plot over the course of the study (figure 5-18B).

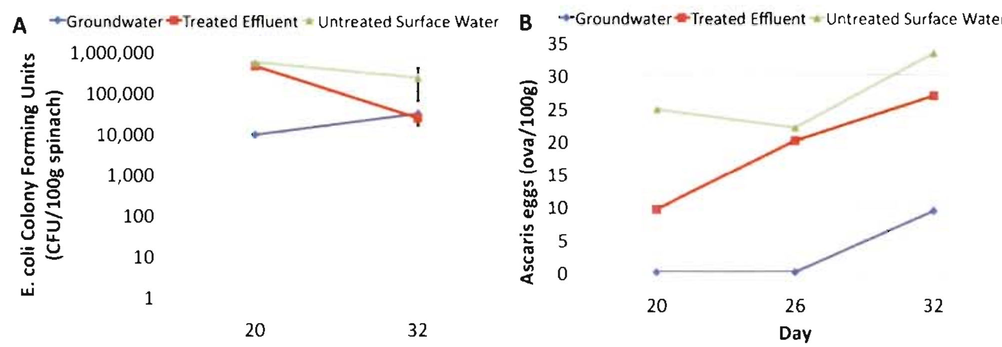


Figure 5-18: *E. coli* (A) and *Ascaris* ova (B) on spinach in this study

Next, recontamination of crops during farmer harvesting will be discussed.

5.5.2.4 Recontamination of Crops

Farmer practices during harvest result in cross-contact between different media (crop, water, and soil). The bundles are wrapped by hand and contact soil often. Then, the bundles are placed under a cloth that has been wetted with nearby water (often wastewater), in order to shade and cool the plants so that they will not wilt (figure 5-19). After observing farmer practices during harvest, a "sanitized" or "clean" harvest was done by this researcher in order to compare a fairly "untouched" crop with those that were harvested by the farmer. Hand sanitizer was

Figure 5-19: After harvest, bundles stacked together and wetted with nearby water (often wastewater) to prevent wilting.

applied to hands, allowed to dry, and care was taken to not touch anything except the crop and sterile sample bag.

Because of the hot temperatures, *E. coli* were expected to die from heat and UV radiation on the leaves of the crop. Using a temperature gun at time of harvest, soil temperatures were found to be as high as 58.3˚C on dry soil, and at the same time, 28.5˚C under leaves in damp areas. Contact with soil, contaminated water, and handling with unwashed hands can recontaminate the crop. The difference was found to be quite large (note the log y axis in figure 21).

Figure 5-20: Sampling at the groundwater plot

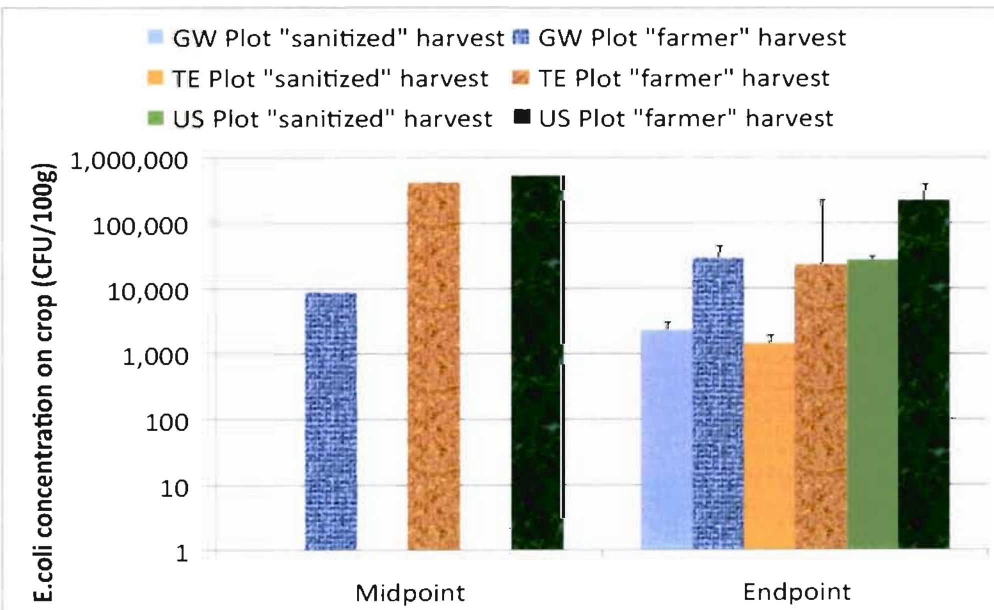


Figure 5-21: *E. coli* found on crop at mid- (farmer harvest) and end-point (both sanitized and farmer) to test for recontamination during farmer harvest. GW: groundwater; TE: treated effluent; US: untreated surface water plots.

For all plots, recontamination during harvest accounted for an increase of one order of magnitude in *E. coli* concentration on spinach.

5.6 Insights and Recommendations for Future Work

The following insights can be made from the results of the site study:

- Consistently large differences in water quality (*E. coli*) yielded relatively small differences in *E. coli* on the crop, when averaged over one growing cycle (this study). Figure 5-22 shows averages for each media (water, soil, spinach) over one growing cycle.

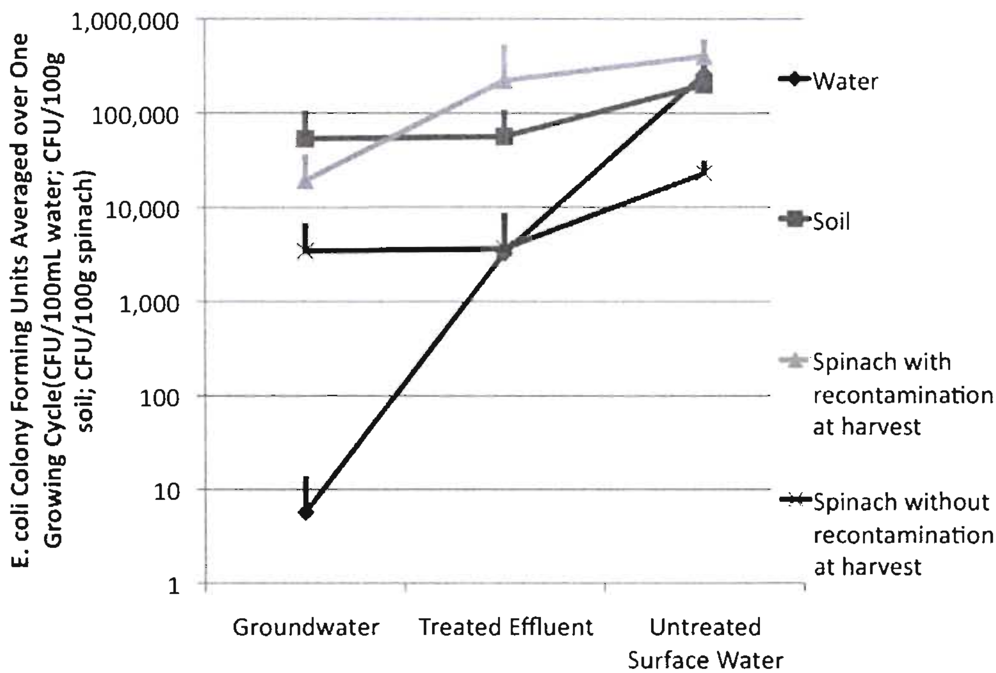


Figure 5-22: *E. coli* averaged over one growing cycle for all media: irrigation water, soil, and crop harvested in two different ways.

- Influencing factors for varying pathogens on crop:
  - Soil contamination was transferred to crop during harvesting practices;
  - Water contamination was transferred to crop during harvest when a wastewater-soaked blanket was used to prevent wilting;

    - The summer heat killed *E. coli* on the surfaces of leaves, but they survived in the soil and in shaded areas;
    - Farmer handling could also be causing cross-contamination between water, soil, and crop.
- Overall, the WWTP removed a high percentage (99%) of the *E. coli* content of water, and therefore, of the irrigation water impact on agriculture. High amounts of nematode ova (*Ascaris*) are able to survive in the soil in extreme conditions and for long periods of time (World Health Organization 2006). Because just one nematode egg can pose a health risk, if ingested, the treated effluent water is still a health threat.

Because this study was done in the dry season, it cannot be directly translated for quantification of impacts throughout the year, especially in the wet (monsoon) season. Dry season conditions could cause higher pathogen concentrations in irrigation water due to minimal dilution from storm water runoff, but high heat could kill bacteria on crop leaves. Furthermore, monsoon conditions are associated with higher amounts of storm water drainage, cross contamination between crop, soil, and water, and lower temperatures. Another study in the wet season is needed to measure these impacts. Furthermore, duplicates done for nematode tests would be useful to determine variation in test results.

Also, further investigation using pathogen die-off models would help to explore *E.coli* die-off and recontamination in this study.

5.7 Sustainability Tradeoffs and Discussion

The sustainability tradeoffs between groundwater use (and related groundwater savings), energy use and greenhouse gas emissions (both energy related GHG and GHG from wastewater), food produced, health risk, and infrastructure cost are shown in figure 5-24. Dollar investments for the required infrastructures for delivering these irrigation waters are noted at the bottom. The five environmental/social/economic sustainability tradeoffs will be compared in a pentagon.

Each parameter was calculated in the following ways. Groundwater use was calculated by using the spinach water requirement, averaged for China, Indonesia, Mexico, and Thailand. These numbers are for one growing cycle of spinach and the average was 3,548 m^3/hectare ± 374 (standard deviation). This converts to approximately 94 gallons/m^2 per month (as the type of spinach/chard in this study grew to harvest in about a month). The amount of water used in the flood irrigation practice (a common practice in this area) is dependent on the type of soil and used 324 gallons/m^2 per month, or almost 3.5 times as much water as the crop needed. Therefore, to accurately quantify groundwater use, the crop requirement was appropriate. Because of the varying units used in the graph, this water use was divided by 50 for all plots in order for it to fit well in figure 5-24.

Energy use and greenhouse gases emitted for the treated effluent and untreated surface water plots were calculated due to the findings in the previous chapter. The total GHG impact from the uncontrolled release in figure 4-11 was used for the untreated surface water plot here, as this irrigation water is released from the city and has only been diluted. For the treated effluent plot, the GHG impact from figure 4-11 for the collected and treated water is used for the treated effluent water here. The GHG emissions per gallon were multiplied by the gallons of water for crop requirement (from the groundwater use calculation above). Additionally, the only energy use for the groundwater plot was for pumping the water. With a pump efficiency of 21%, this energy use was estimated over 1 year. The total GHG emissions for each plot were multiplied by 10 for figure 5-24.

Because results are shown as larger being worse, food production is calculated as the inverse of production. The amount produced in each plot here is simply the inverse of the crop quantities described earlier. This amount was multiplied for 10 growing cycles. Then, the totals for each plot were multiplied by 50 for figure 5-24.

The probability of disease was calculated for ingesting *E. coli* when eating 100g of farmer-harvested crop per week for a total one year. The percent probability is shown in figure 5-24. Because *Ascaris* can survive for long periods in the soil, the amount seen on the crop is thought to have been transferred from the soil because zero nematodes were found in the irrigation water. See figure 5-23 for all health risk results from this study.

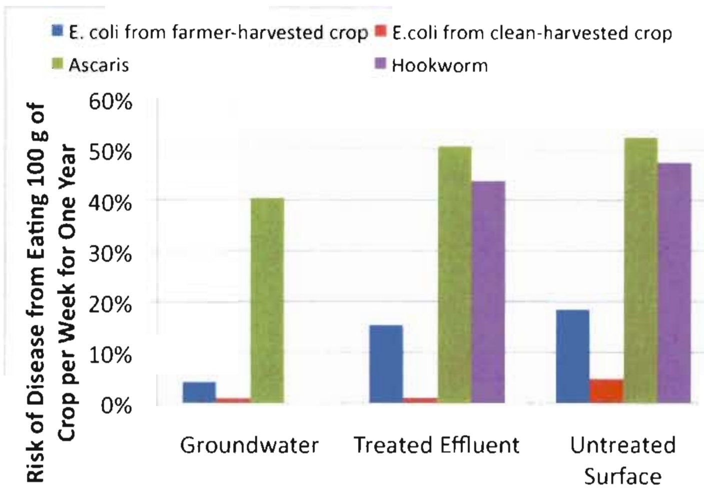


Figure 5-23: Probability of contracting a disease from ingesting 100g of raw spinach per week for one year

In table 5-7, all agriculture plots show statistically significant differences between farmer-harvested and sanitized-harvested crop. Also, with farmer-harvested crops, treated effluent and groundwater plots were similar, while the untreated surface water plot was slightly more contaminated. The same was seen with the sanitized harvest with all measurements one order of magnitude lower than their plot-counterpart (see figure 5-21).

Table 5-7: Statistical tests of E.coli on crop at time of harvest

Study Period	Hypothesis for crop grown with noted irrigation waters		Outcome	Details	
Harvest Farmer Handling from All Three Plots	Untreated is similar to WWTP		Reject	p<0.025	df=8
	WWTP is similar to groundwater		Fail to Reject	p>0.1	df=2
Reharvest Farmer Handling from All Three Plots	Untreated is similar to WWTP		Reject	p<0.05	df=8
	WWTP is similar to groundwater		Fail to Reject	p>0.1	df=5
Reharvest Farmer Handling versus Sanitized Handling Plot by Plot	Farmer handled spinach was similar to sanitized harvest spinach from plot:	Untreated	Reject	p<0.025	df=6
		Treated*	Reject	p<0.1	df=3
		Groundwater	Reject	p<0.1	df=4

* Data from harvest of site study is used here because tests were uncountable at reharvest

The total capital costs for installing a groundwater well is $6,000 USD based on a project in Africa (The Water Project 2011). The useful life of the well structure and pump is assumed to be 10 years on average. Therefore, $600 USD per year for a groundwater well is the investment to drill a well, pour a concrete base, and buy and install a pump mechanism. The infrastructure capital costs for NWWTP was $2,912,785 USD (12 crore Indian Rupees in 2007 at an average conversion rate of 41.2 INR to 1 USD (x-rates 2011). The yearly cost of this WWTP is $97,093 (2007 USD). Operating costs are not included here.

From figure 5-24, it is clear that the use of treated effluent and untreated surface water for urban agriculture contributes to higher energy use/greenhouse gas emissions and health risk. However, using groundwater for urban agriculture contributed harmful impacts towards the other two factors, groundwater use and food production. It is important to note that if fertilizer were used to increase productivity, GHG emissions would increase.

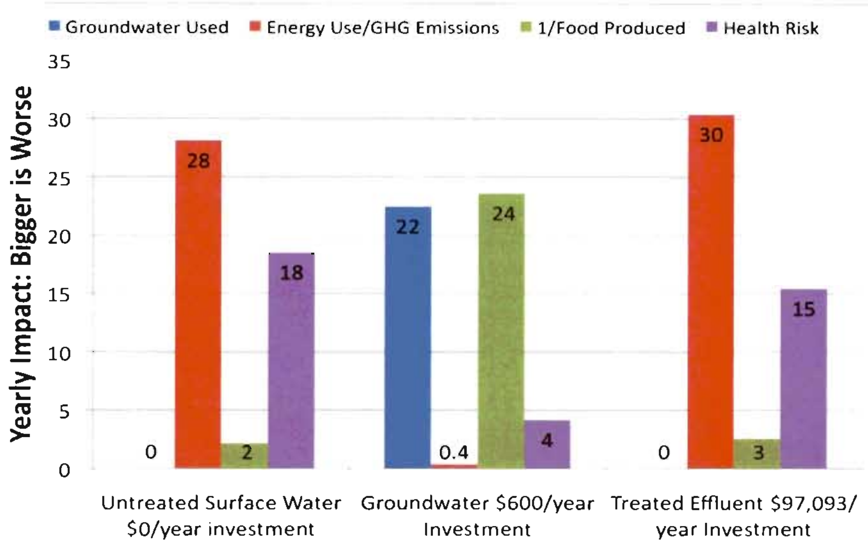


Figure 5-24: Tradeoffs between groundwater use, energy and GHG, 1/food produced, and health risk in this case study. Units were normalized as noted above.

The sustainability pentagon is shown in figure 5-25. All axes range from 0% (best) to 100% (worst). For each plot, every parameter is compared to the "worst" plot as a percentage.

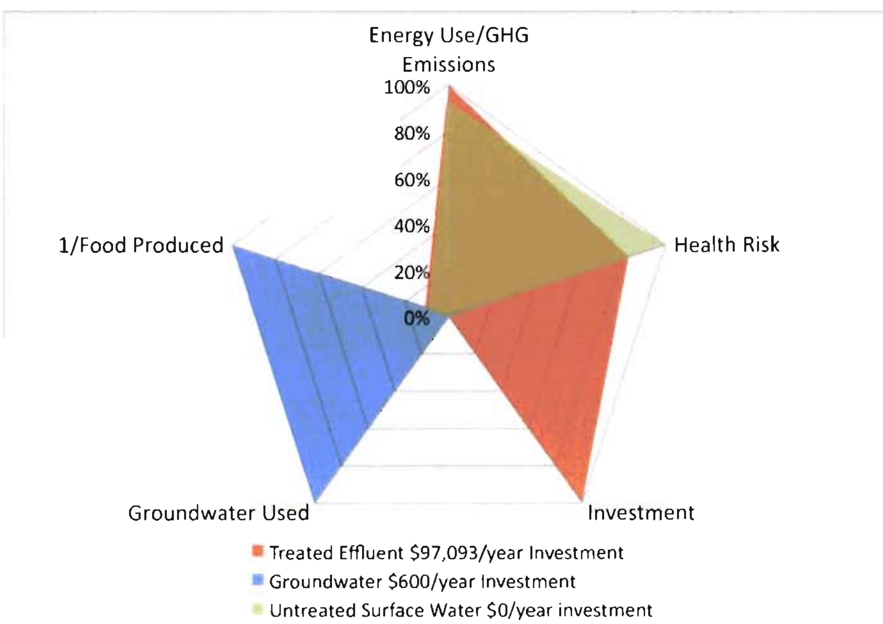


Figure 5-25: Sustainability pentagon showing tradeoffs between energy use/GHG emissions, health risk, infrastructure capital investment, groundwater use, and the inverse of food produced. All sites are compared to one another as a percentage (the "worst" site being 100% for each parameter).

Use of treated effluent and untreated surface water for urban agriculture were similar in terms of greenhouse gas emissions and health risk. Even though the WWTP removed 99% of *E.coli* from water, the crop was still contaminated only slightly less than untreated water.

While this WWTP did not provide as many benefits to urban agriculture as presumed, there may be benefits towards cleaning up the Musi River and avoiding groundwater contamination, that are beyond the scope of this work. Use of groundwater for urban agriculture contributed harmful impacts towards the other two factors: groundwater use and food production. It is important to note that if fertilizer were used to increase productivity, GHG emissions for the groundwater plot would increase. Importantly, falling groundwater levels are a major issue.

It is clear that choices made based on these tradeoffs will be different for different situations and groups of people. For example, for a farmer who doesn't have money to invest in a groundwater well and for whom obtaining water from a WWTP is difficult, the sole choice would be to use untreated surface water. This choice will bring the most income, but also the most health risk.

However, for an agency that wants to reduce GHG emissions and health risk, they may choose to fund groundwater wells for urban farmers. This choice may work well in a city with a large amount of groundwater resources, but many cities in India are experiencing drops in their groundwater tables. For this case, additional fertilizer (commercial or biosolids) will be necessary to provide nutrients for crops.

The Government of India has chosen to build WWTPs in order to clean up rivers around the country. While there are benefits provided by WWTPs, effluent released back into a polluted riverine system may not be worth the investments in energy or money. Because at this time in Hyderabad, India, 1.4 million properties are not connected to the sewer system, and 0.9 million properties do not have access to toilets, these uncontrolled wastewater releases will overwhelm the 99% pathogen reduction of only 40% of produced wastewater. For *Ascaris*, much higher removals are needed because just one egg can infect a person.

While the purpose of WWTP implementation is not specifically to provide irrigation water to urban farmers, farmers could benefit from treated water for use on their crops. Other WWTP technologies or natural treatment of wastewater, such as natural buffer strips along rivers, may be better solutions for this coupled system.

Social actors should be queried as to how they would deal with falling groundwater levels and other issues related to this study. Future work is needed on how social actors would weigh these tradeoffs.

6. Conclusion

6.1 Major Contributions and Conclusions

The major contributions of this work are to four areas of literature. First, the data analysis from chapter 1 contributes to the urban metabolism literature by examining energy use and intensity in water/wastewater infrastructures in developing country cities. It is the first study to examine benchmark data for waste and wastewater infrastructure for the developing world. From this analysis, water and wastewater sector were found to contribute a large proportion to community-wide GHG emissions. When process emissions are included, the water/wastewater sector can be a large contributor at approximately 6-32% of community-wide GHG emissions. This proportion was shown to be higher for Indian cities than for US cities. On a per gallon basis, there was more electricity use for water supply when compared to that for wastewater in India. This comparison was the opposite for US cities. On average, for Indian cities included in this study, only 48% of generated wastewater was treated.

This study also makes contributions to the WWTP LCA literature by conducting a first LCA using operating data from an Indian WWTP with consequential impacts from water reuse in urban agriculture. The WWTP LCA showed that electricity use was relatively small compared to on-site process emissions and embodied energy, as electricity use is normally much higher than these in US WWTPs. A method was developed method for consequential LCA of urban agriculture using the DAYCENT model. Results showed that high water flows and land constraints limited nutrient cycling in urban agriculture.

The farming site study contributes to the urban agriculture literature by completing a first field study of pathogen impacts from treated and untreated wastewater use for irrigation. This study showed that even though irrigation water quality was very different, the crop quality was fairly similar.

Figure 6-1: Farmer, Chandriah, and family helping with harvest at the groundwater plot

Recontamination from farmer handling contributed about one order of magnitude in concentrations on spinach leaves. This is also a first urban agriculture case study that has tried to quantify tradeoffs.

Overall, an integrated sustainability assessment to quantify tradeoffs of WWTP effluent reuse in urban agriculture linked water/wastewater, energy, infrastructure capital investments, crop production, and health. This sustainability pentagon demonstrated that quantitative tradeoffs can be computed and visualized. This could be a useful tool for decision making by social actors.

6.2 Recommendations for Future Work

In the urban metabolism study, further data gathering and examination for water supply sources, distances pumped, and drinking water and wastewater treatment technologies could elucidate the reasons for the variation in energy use between cities for water and wastewater infrastructures. For the WWTP LCA, much more could be learned from on-site CH_4 and N_2O measurements from both the anaerobic and aerobic treatment processes. Also, forthcoming data from NWWTP and other WWTPs in Hyderabad is expected to further the understanding of both on-site emissions and off-site emissions following effluent release to the riverine system. For the urban agriculture field study, more samples for *Ascaris* on crop and in soil would be useful to better quanitfy the variation and extent of contamination. A similar field study in different seasons, especially the wet (monsoon) season, would be useful to assess crop pathogen content throughout the year.

Overall, benefits to urban agriculture may be better realized from other methods of wastewater treatment. Assessing the potential of natural treatment/vegetative buffer strips for megacities where the majority of wastewater is untreated, or alternatives to flush toilets leading to centralized WWTPs, could be more favorable from the perspective of water reuse for urban agriculture.

APPENDIX A: DAYCENT METHODOLOGY

The DAYCENT model uses a daily timestep and evolved from the CENTURY model, which has a monthly timestep. These models were generated to more accurately quantify N_2O emissions from fields. As described by Dr. Parton at CSU, most studies show that 1-2% of total nitrogen is released as N_2O and this is not specified whether this is directly from the field or leached. Results from CENTURY and DAYCENT generally show that less than 1% of total N emitted from agriculture practices. More N_2O is seen from sandy soils versus clay soils because clay soils absorb N_2O. The models predicts the N_2O/NO_x ratio. These models are based on a simple ecosystem model which includes long term changes in soil organic matter, nutrient cycle and related plant production, and hydrology. he C:N ratio affects the nitrogen mobility in the model: a C:N ratio of 80 takes about 5 years for all nitrogen to be mobilized, while nitrogen mobilization is almost immediate with a C:N ratio of 20. The same phenomena is seen for phosphorus.

Input data for the crop included to following parameters: annual vs. perennial, the amount of carbon in the plant, C:N ratio, and the temperature curve for which mimics actual growth production. Events were scheduled in blocks, or sets of events, that can repeat on a yearly basis. Nitrogen and organic matter multipliers were added to imitate historical inputs. The field practices were defined, such as grazing, burning, plowing, cultivating, harvesting, etc. Trees can be added to the model, but were not used for this study. The layers of soil were defined by nutrients and physical characteristics as determined in Chapter 5.

Once the schedule file is made, the model is run with the command prompt function in a Windows operating system.

APPENDIX B: PATHOGEN TEST METHODOLOGY

1) Methodology for *E. coli* and Total Coliform Quantification by Hach Membrane Filtration in Water, Vegetable, and Soil

Adapted from Hach method, IWMI protocol for this test, and L. Miller's experience.

Equipment and reagents:

Petri dishes with Pad
Gridded Filter Papers
mColiBlue Broth
Hand Sanitizer
Plastic bags
Dilution Water Concentrate for Buffered Saline for washing spinach
Pipet tips
100 ml beakers for dilution
Membrane Filtration Unit
10mL Sterile Pipets
1L glass bottles for buffered dilution water or distilled water
Knife for harvesting spinach
Vacuum pump with tubing
Laminar flow hood & Gas connection
Incubator set at 35 C
Digital scale
Forceps
100 ml measuring cylinder
At least 10 1L glass bottles for buffered dilution water and distilled water
Pipettor
Distilled Water
Tip box
Ice packs
Styrofoam cooler

Before going to field

1. Sterilize all appropriate materials the day before
2. Weigh, label, and record empty, sterile, sample bags for vegetable samples

Sampling Water

3. Gather a composite sample in a large, clean sampling can
4. Mix well and pour a 100mL sample into a small, sterile sampling bag or glass bottle
5. Be sure to close sampling vessels completely and securely
6. Store sample bags in a ice box as soon as possible
7. Return to the lab and start experiment as soon as possible (same day, within 2-3 hours)

Sampling Soil

8. Gather a soil from 3 random spots in the plot (first carefully removing top 3cm of soil before collecting sample) and place samples together in a sterile sampling bag.
9. Be sure to close sampling bag completely and securely
10. Store sample bags in a ice box as soon as possible
11. Return to the lab and start experiment as soon as possible (same day, within 2-3 hours)
12. Weigh 10g of soil in a sterile flask.
13. Add 95mL of 0.8% NaCl solution
14. Place on a shaker for 60 minutes
15. Filter through a clean (sterile, if possible) strainer
16. Discard the solid portion and the liquid portion will be used to make dilutions.

Sampling Vegetable

17. Take a grab sample of approximately 100g vegetable (was about 6 bundles for spinach, each bundle is approximately 50g each) at harvest from three random spots in the 3 different plots
18. Place samples (about 2 bundles each) in sterile ziplock bags and label for plot
19. For sanitized harvest only:
    a. Squeeze small amount of hand sanitizer on knife or scissors and spread evenly over entire surface and let dry
    b. Squeeze small amount of hand sanitizer on hands and spread evenly over entire surface by rubbing hands together and let dry
    c. Harvest 100g spinach with minimal contact with soil or anything that is not sterilized
    d. Immediately place into a sterile bag without touching soil
20. Be sure to close bags completely with minimal air left inside
21. Store sample bags in a ice box as soon as possible
22. Return to the lab and start experiment as soon as possible (same day, within 2-3 hours)
23. Plan dilutions in notebook (1:10, 1:100, 1:1,000, 1:10,000, 1:100,000, 1:1,000,000)
24. At a scale, use sanitizer on hands, let dry, and carefully untie any tied bundles in bag with as little contact as possible and re-close bag
25. Weigh each sample in bag and record weights.
26. Add 4 times volume (milliliters) phosphate buffered saline to weight (grams) spinach
    a. 400mL to 100g spinach
27. Close bag tightly
28. Shake spinach vigorously with buffer solution
29. Cut the zip lock bag in a corner and carefully pour buffer into sterilized beaker (inside sterile hood if possible)

The following steps were used for water, soil, and vegetable samples:

30. De-contaminate laminar airflow with 70% ethanol
31. Make 100mL dilutions appropriate for samples, using sterile distilled water (higher than 1:1,000 for soil and 1:25 for spinach can clog filter paper during filtration)
32. Label sterilized Petri plates according to plan
33. Assemble the filter assembly and light the flame
34. Flame forceps to sterilize them
35. Place filter membrane on the filter holder with forceps
36. Screw in the funnel so that water will not leak (over tightening of the funnel can tear the filter.)
37. Before filtering, mix 100 mL sample properly (which avoids cluster colonies).
38. Lift the funnel cover aseptically and pour the sample into the funnel
39. Cover the funnel immediately. Connect the vacuum pump to the filter assembly, using appropriate tubing
40. Open plastic broth ampule and pour broth onto absorbent pad
41. Place the filter membrane on broth added absorbent pad in a petridish by using sterile forceps (flame forceps each time before using them)
    a. Use rolling motion to avoid trapping air bubbles between filter and pad, then cover carefully with lid
42. Rinse the filter holder funnel by pouring sterile water into the funnel to completely filter any sample
43. Incubate the petri dishes at 35°C overnight and check in morning if the colors have developed properly and if the pad isn't dried out
    a. If pad is starting to dry, read results immediately
    b. If pad is wet and color has not yet developed, let plates incubate for the remaining of 24 hours since initially placed into incubator, then read results
44. Read under a magnifying glass, if necessary.
45. Red and purple colonies are counted as *E.coli* and blue colonies are total coliforms. *E.coli* was the focus of this test and the most accurate counts are from plates with 20-80 *E.coli* CFU/plate.
46. Make calculations according to dilutions and weights of vegetable and soil samples.

2a) Methodology for Ascaris and Hookworm Quantification in Water and Vegetable samples

Adapted from IWMI method as conveyed by Dr. Priyanie Amerasinghe (Amerasinghe, Weckenbrock et al. 2008).

a. **Water**
    1. Gather 5-10L of water as a composite sample
    2. Transfer to the lab and allow samples to settle undisturbed.

3. If enough wide mouthed clear flasks or beakers are not available to settle all of the collected liquid, after 3-4 hours, carefully discard the upper layers of water (if the samples bottles are easy to work with and clear, no transfer is necessary).
4. Transfer all of the remaining liquid to the wide-mouthed glassware and rinse with a minimum of detergent solution (1% triton) and include the washing liquid in the sample.

b. **Vegetable**

5. Weigh 100 g of vegetable sample, put into the sterile Zip lock bag
6. Pour 400ml phosphate buffer solution and wash vigorously
7. Combined 3 samples together and consider as 1 sample (one sample pellet is too little and after combining samples, the pellet was 0.5ml).
8. Cut the Zip lock bag in a corner with scissors and pour liquid into a wide mouth glass beaker.

The following steps were used for both water and vegetable samples:

9. Allow to settle undisturbed overnight.
10. Next day, carefully remove the supernatant without disturbing the settled sediment by gravity flow or suctioning.
11. Carefully transfer the sediment into the centrifuge tubes with pipet. Rinse the container well with a minimum of detergent solution (1% triton) and include the washing liquid in the sample.
12. Centrifuge at 1000g for 10 min. Carefully pipette out the supernatant, keeping the pellet.
13. Suspend the pellet in equal volume of aceto-acetic buffer (pH 4.5).
14. If the pellet is less than 0.5ml, add 1ml of buffer; if the pellet is 2ml, add 2 ml of buffer (1:1ratio)
15. To the same tube, add two volumes of Ethyl acetate (If the pellet is 2ml add 4ml of ethyl acetate).
16. Mix the solution thoroughly by using a vortex mixer, until the pellet is dislodged and an even suspension is seen.
17. Centrifuge the sample at 1000g for 15 minutes.
18. The sample will now have 3 distinct phases. All the non-fatty heavier debris including helminthes eggs and protozoa at the bottom layer. Above this will be the buffer. The rest will be top layer, which may appear as a dark plug.
19. Discard the 2 top layers and record the volume of the pellet.
20. Add zinc sulphate (specific gravity 1:18) to the pellet, at the ratio of 1:5 (if the pellet is 1ml, add 5ml of zinc sulphate solution.)
21. Mix the sample thoroughly by pipetting up and down.
22. Transfer the sample to a Mc Master slide using a pipette (The volume of Mc Master slide is 0.3ml).
23. Count all the eggs seen within the grid in both chambers of the Mc Master slide (Total volume =0.3ml, each chamber has 0.15ml).
24. Repeat this count until all liquid is gone.

25. Same method is followed for each sample.
26. Calculate the number of eggs per liter from the equation:

$$N = \frac{AX}{PV}$$

where: N= number of eggs per liter water (or equivalent weight of vegetable sample)
A= number of eggs counted in the McMaster slide of the mean of counts from all slides
X= volume of the final volume in the tube processed (mL)
P= volume of the McMaster slide (0.3mL)
V= original sample volume (Liters)

2b) Methodology for Ascaris and Hookworm Quantification in Soil Samples

Adapted from Zenner et al, 2002, "A standardized method for detecting parasite eggs and oocysts in soils" (Zenner, Gounel et al. 2002).

1. Saturated magnesium sulphate solution was used as the flotation solution (Dissolve $MgSO_4$ in distilled water while in a boiling water bath until completely saturated).
2. 50g of soil added to 200ml water
3. Mix thoroughly with metal rod for 20 seconds
4. Filter through coarse sieve (pore size 0.1mm) to remove large size debris.
5. Mix to homogenize again and transfer to 4-50mL centrifuge tubes
6. Centrifuge at 150g for 5 minutes
7. Randomly select two tubes
8. Discard supernatant
9. Resuspend each pellet in 30 mL of saturated magnesium sulphate
10. Divide each into 2-15mL centrifuge tubes (4 total) and fill until have positive meniscus
11. Cover each with an 18x18 mm coverslip
12. Centrifuge at 150 g for 5 minutes
13. Remove coverslips, placed on a slide, and examined microscopically
14. Record the number of eggs adhering to each coverslip
15. Scrape the inside of each tube with a wire (or thin side of metal rod) in order to re-suspend any eggs which had adhered to its walls
16. Top-off tubes with flotation media, cover, centrifuge, and examine the coverslips as before
17. Repeat the coverslip recovery and count process five times
18. Tally the number of eggs after each centrifugation and total
19. Calculate the number of eggs per 100g soil from the equation:

$$N = \frac{A}{(12.5g * 8)}$$

where: N= number of eggs per 100g of soil sample

A= the sum of all eggs counted on all coverslips for that sample

100g soil= 12.5g soil * 8

APPENDIX C: ABBREVIATIONS

BOD_5	Biochemical Oxygen Demand 5-day
COD	Chemical Oxygen Demand
CH_4	Methane (gas)
DO	Dissolved Oxygen
DT	Detention Time
FC	Fecal Coliforms
HMWSSB	Hyderabad Municipal Water Supply and Sewerage Board
HRT	Hydraulic Retention Time
kWh	Kilowatt Hours
MG	Million US Gallons
MPN	Most Probable Number
mt	metric tons
N_2O	Nitrous Oxide (gas)
NWWTP	Nallacheruvu Wastewater Treatment Plant
SRT	Solids Retention Time
TC	Total Coliforms
TSS	Total Suspended Solids
VSS	Volatile Suspended Soilds
WWTP	Wastewater Treatment Plant
W/WW	Water and Wastewater

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