ASSESSING LIFECYCLE COSTS AND BENEFITS OF INTEGRATED SANITATION AND ENERGY RECOVERY STRATEGIES IN LOW-INCOME COUNTRIES

A DISSERTATION
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Abstract

By mid-century, the global urban population is projected to increase by 2.4 billion people with 60% of growth expected to take place in cites of sub-Saharan Africa (SSA) and South Asia (SA) (UN-DESA, 2015). These regions have some of the lowest rates of urban sanitation coverage and modern fuel access, with only 41% and 67% estimated to have access to improved sanitation and 19% and 55% estimated to use modern fuels for cooking in SSA and SA, respectively (WHO/UNICEF, 2015a; Legros, et al. 2009).

In recent years, decentralized sewered and non-sewered sanitation management approaches have gained traction in low-income SSA and SA cities as alternatives to expensive and resource intensive conventional wastewater treatment. Energy recovery from domestic waste has also received increasing attention as a strategy to offset operational costs of sanitation services, provide alternative energy resources and reduce environmental impacts of waste treatment systems. Waste-to-energy technologies may simultaneously contribute to multiple Sustainable Development Goals (SDGs) including achieving universal access to safely managed sanitation; universal access to modern, clean fuels; and integration of climate change measures into national policies, strategies and planning including fostering low emissions development (UN, 2015). However, despite increasing adoption of alternative sewered and non-sewered sanitation approaches and growing interest in energy recovery, little empirical evidence exists regarding the extent to which energy recovery may offset costs of sanitation systems and the long-term resource requirements, environmental impacts and financial costs and benefits of alternative sanitation approaches.

The objectives of this dissertation are to (1) provide empirical evidence on the costs and benefits of integrated sanitation and energy recovery strategies in low-income countries, (2) develop a lifecycle based framework to enable sanitation planners to systematically investigate the resource use, environmental impacts and financial costs and benefits of decentralized sewered and non-sewered sanitation approaches and (3) demonstrate application of this framework using primary data from operational

sanitation systems in Zambia and India, thus also generating the first comprehensive analyses of decentralized sewered and non-sewered sanitation management in urban SA and SSA.

Chapter 2 uses primary data and a cross-sectional study of 40 households in Arusha, Tanzania to investigate the impacts of domestic, agricultural anaerobic digester (AD) use. Findings suggest that AD adoption has the potential to reduce fuel-wood use, energy-related expenditures, and time-costs of energy procurement; to lower CO_2e emissions; and to increase farm incomes. Domestic AD investments were also found to have a positive net present value across a wide range of discount rates.

Chapter 3 evaluates the treatment efficiency, energy production, GHG emissions and financial costs and benefits of communal ADs using primary data collected in 3 peri-urban Zambian communities. Results from a 6-month monitoring period of 15 ADs and a cross-sectional survey of 120 households suggest that ADs have the potential to offer greater GHG emission reductions and financial benefits relative to conventionally used septic tanks (STs). Water access, baseline fuel use and spatial optimization of biogas supply and demand were identified as key conditions affecting performance of ADs relative to conventionally used STs.

Chapter 4 develops a lifecycle framework to quantify water, energy, global warming potential (GWP) and financial costs and benefits of decentralized sewered and non-sewered sanitation approaches. Application of this framework is demonstrated using primary data collected from four operational sanitation systems located in Zambia and India. Findings from this comparative study suggest (1) the percent of biogas recovery and baseline fuel use strongly influences lifecycle energy use and GWP for sewered systems, whereas biogas recovery had little influence on these parameters for non-sewered systems; (2) capital infrastructure and water supply are important drivers of lifecycle energy use; (3) direct methane emissions drive the magnitude of GWP impacts; and (4) sewered systems have significantly higher user and agency costs *versus* non-sewered systems; however, non-sewered system users may incur higher upfront capital costs and a greater proportion of lifecycle costs compared to sewered system users.

Chapter 5 discusses some of the main conclusions from this dissertation including (1) integrated sanitation and energy strategies have the potential to offer significant sanitation, energy and climate co-benefits, with greater benefits accruing under certain conditions; (2) different financial models and greater incentives are needed to attract investment in emission reduction approaches; and (3) valuable insights can be gained on the costs, benefits, tradeoffs and key areas for system improvements by using a lifecycle assessment framework to evaluate different sanitation approaches. Expanded system boundaries, including water provision and interactions between recovered and conventional resources, are critical to understanding full lifecycle costs. Chapter 5 further discusses the major contributions of this dissertation, implications for sanitation planning and recommendations for future work.

Acknowledgements

As I reflect on these last several years, I realize the crux of this PhD is not the end-product of this dissertation, but rather the journey to get here. The polished text and tables of a dissertation mask the circuitous path of a PhD, particularly for one in which I have been fortunate enough to live, explore and learn on three continents. I am humbled by and forever grateful to the many people who have encouraged and supported me along the way. My life has been enriched and forever changed by all of you.

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1 Introduction

The first chapter of this dissertation outlines current sanitation, energy access and urbanization challenges in low-income countries; describes adaptation and mitigation links between sanitation and climate change; provides an overview of global sanitation targets and feasible infrastructure options; and describes the potential for integrated sanitation and energy recovery systems. Research questions explored in this dissertation are provided and subsequent chapters are outlined.

1.1 Current sanitation and energy challenges

1.1.1 Sanitation, energy access and urbanization

Access to basic services such as sanitation and clean energy remains a persistent challenge throughout the developing world. An estimated 2.4 billion people worldwide do not have access to hygienic sanitation and 3 billion people use biomass or coal fuels for cooking energy (WHO/UNICEF, 2015a; WHO, 2016). The associated health impacts are considerable, with a combined 4.7 million premature deaths globally each year attributed to inadequate sanitation (0.4 million) and poor indoor air quality associated with clean energy access (4.3 million) (Pruss-Ustun et al., 2014; WHO, 2016). These figures are particularly pronounced in sub-Saharan Africa (SSA) and South Asia (SA) where only 30% and 47%, respectively, of the population uses improved sanitation and 19% and 55%, respectively, use modern fuels for cooking (WHO/UNICEF, 2015a; Legros et al., 2009).

The challenge of sanitation and energy access in SSA and SA is compounded by unprecedented rates of growth in these regions, particularly in SSA where the population is projected to double in the next thirty years (UN-DESA, 2015). Moreover, a considerable amount of growth is expected to take place in cities: by mid-century, the global urban population is projected to increase by 2.4 billion people with 60% of growth expected to take place in SSA and SA (UN-DESA, 2015).

SSA is also the only region in which the number of poor people has increased over the past several decades. Currently, half the region's population lives in extreme poverty (World Bank, 2013). Such rapid growth without concomitant gains in economic development poses enormous challenges for city governments, faced with increasing demand for public service provision against very limited resources (Beard et al., 2016). Among urban infrastructure systems, sanitation has particular challenges. Like water supply, it is essential for public health. Unlike water supply, effective demand (willingness and ability to pay) for sanitation among households is particularly low (Whittington et al., 2000).

1.1.2 Sanitation and climate change

Decisions on sanitation infrastructure have implications from both a climate change mitigation as well as adaptation perspective. Sanitation infrastructure produces considerable greenhouse gas (GHG) emissions, accounting for 1.5% of global anthropogenic emissions from wastewater treatment and 1-2% of global anthropogenic methane (CH_4) emissions from pit latrines (Herzog, 2005; Reid et al., 2014).

On the other hand, increasingly variable climatic conditions and water scarcity are likely to further intensify sanitation service challenges within the next century (Howard et al., 2010; Sherpa et al., 2014). Global climate change is projected to decrease water availability in already water scarce regions, while increasing precipitation and flooding risk in wet regions (IPCC, 2014). Both conditions adversely affect sanitation: conventional sewerage and wastewater treatment is prone to operational failure without adequate water and energy, while flooding in densely populated urban slums with on-site sanitation (i.e. pit latrines) can result in widespread fecal contamination and an associated burden of diarrheal disease.

1.1.3 International development targets and sanitation infrastructure

The Millennium Development Goals (MDGs), established in the year 2000, outlined an international commitment by all United Nations member states to eradicate extreme poverty by the year 2015. The MDGs defined 'improved sanitation' based solely on access to a toilet facility (WHO/UNICEF, 2015b). Recognizing that effective treatment and safe disposal of excreta are necessary to achieve desired health gains of sanitation interventions, the Sustainable Development Goals (SDGs), which establish development targets between the years 2015 – 30, propose 'safely managed sanitation services' as the new target indicator (WHO/UNICEF, 2015b). Specifically, this target is defined as "use of an improved sanitation facility which is not shared with other households and where excreta are safely disposed in situ or transported and treated off-site" (WHO/UNICEF, 2015b). The SDG sanitation target aims to achieve universal access to safely managed sanitation by 2030 (UN, 2015). In line with this expanded scope, a greater proportion of investment in the coming decades will be directed to the backend of the sanitation service chain (SSC), including safe and systematic containment, collection, conveyance, treatment and disposal or reuse.

Major infrastructure investments will be necessary to meet the SDG target of universal access for rapidly growing populations in SSA and SA. Conventional sewered networks will not be feasible for the vast majority of SSA's urban residents in the foreseeable future given high capital and operational costs, as well as substantial water and energy resource demands, which are often limited and unreliable in these settings (Corominos et al., 2013). Most urban households will instead continue to use on-site sanitation facilities such as pit-latrines, with financing, construction, and maintenance being the responsibility of users themselves (UNICEF/WHO, 2014; Blackett et al., 2014).

In recent years, non-sewered and decentralized sewered approaches have received attention as alternatives that can help expand excreta management options for developing country cities. Non-sewered approaches are based on the principle of organizing a system for the "safe and systematic collection, transport, treatment and disposal/reuse of raw or partially digested slurry or solids" (Tilley et al., 2014). Non-sewered systems do not require communal sewer networks, and thus minimize the volume of water required for conveyance. By contrast, decentralized sewered systems move wastes through sewers to local small-scale treatment facilities. As such, they typically require considerably less water and energy inputs as compared to conventional sewerage and wastewater treatment (Gutterer et al., 2009). Moreover, because they generate concentrated waste streams with lower water content, both systems can be designed to recover energy, nutrients and water from domestic wastes. While these alternative approaches have traditionally been viewed as temporary or 'stop-gap' solutions, they are increasingly considered to be among the few viable long-term sanitation approaches for cities in low-income regions (Peal et al., 2014).

1.1.4 Sanitation and resource recovery

There is a growing recognition that the objectives of wastewater treatment must advance beyond public health and pollutant removal, and endeavor to work towards principles of ecological sustainability including recovery of resources from waste streams (Corcoran et al., 2010; Corominos et al., 2013). This shift in thinking is reflected in the SDGs, which now include "substantially increasing recycling and safe reuse of wastewater" as a sanitation target. The changing paradigm of waste as a resource rather than a costly problem creates opportunities for simultaneously addressing sanitation and resource challenges with a single approach. Recovering resources from waste, including energy, water and nutrients, may offset operational costs of sanitation services, minimize environmental impacts and provide alternative resource options (Guest et al., 2009; McCarty et al., 2011; Cornejo et al., 2013). Moreover, resource recovery may be particularly beneficial in low-income countries where limited public financing is available for initial or recurrent costs of sanitation infrastructure and low willingness to pay for wastewater treatment contributes to poor sustainability of services (Whittington et al., 2000).

Energy recovery in the form of biogas produced *via* anaerobic digesters (AD) is one example of a waste treatment strategy that provides additional energy co-benefits. Anaerobic digestion technologies of varying complexity, arrangement and scale have been used throughout the world to treat organic wastes and generate energy. As organic waste biologically decomposes under anaerobic conditions, biogas, composed primarily of methane (CH_4), carbon dioxide (CO_2) and nitrogen (N_2) is produced

(Rittman and McCarty, 2001). The CH_4 content of biogas can be recovered and used as an energy source.

Large-scale anaerobic digesters (AD) are commonly used at centralized wastewater treatment plants in industrialized countries to treat bio-solids produced by municipal wastewater treatment, with recovered CH_4 generally converted to electricity (Rittman and McCarty, 2001). Various centralized anaerobic treatment technologies have also been used for the direct treatment of dilute, domestic wastewater in middle-income countries in tropical climates, particularly in Latin America (Seghezzo et al., 1998).

Household-scale ADs have been used in rural areas of developing countries for domestic energy production, particularly in China and India where government initiatives have supported the installation of an estimated 43 million and 4.75 million, respectively (Bond and Templeton, 2011; REN21, 2016). More recently, several programs have been initiated that aim to increase the number of AD installations in SSA (ABPP, 2012; TDBP, 2012).[1] In such installations, livestock manure is generally utilized as the main organic waste. The decomposed bio-slurry discharged from the AD can also be used as a soil amendment and nutrient resource. Household-scale ADs operate based on similar anaerobic processes as large-scale systems, but are comparatively less complex, generally requiring no external energy inputs and limited monitoring of biological processes. Household facilities are also generally designed so that biogas can be used directly rather than converting it to electricity. A limited pool of peer-reviewed literature documents the performance, costs and benefits of household ADs in developing countries (Chen et al., 2010; Van Groenendaal and Gehua, 2010; Laramee and Davis, 2013). Comparatively fewer communal ADs have

[1] Substantial efforts to increase bio-digester installations worldwide are also ongoing: an estimated 22 million new bio-digester plants were installed in China between 2006 and 2011 (REN21 Secretariat, 2011). Major biogas programs are also underway in Vietnam, Nepal, and India, where a total of 105,000 bio-digesters were installed in 2010 (REN21 Secretariat, 2011). Additionally, the Africa Biogas Partnership Programme (ABPP) aims to install 70,000 domestic bio-digesters in six sub-Saharan African countries over the 5-year period ending in 2013 (ABPP, 2012).

been implemented for the treatment of domestic wastewater at the neighborhood or community level. Correspondingly, little information is available characterizing treatment performance, energy production potential or financial viability of ADs at the community scale (Reynaud, 2014; Tilmans et al., 2014).

Waste-to-energy technologies, such as ADs, may simultaneously contribute to multiple global development agendas. For example, within the SDGs, ADs may contribute towards goal 6, which aims to achieve universal access to safely managed sanitation and halve the proportion of untreated wastewater globally; goal 7, which aims to achieve universal access to modern, clean fuels and substantially increase the share of renewable energy in the global energy mix; and goal 13, which includes integration of climate change measures into national policies, strategies and planning including fostering low emissions development (UN, 2015).

1.2 Knowledge gaps and research questions

Despite the increasing adoption of alternative sewered and non-sewered sanitation approaches and growing interest in resource recovery, little empirical evidence exists regarding the long-term resource requirements, environmental impacts and financial costs and benefits of new alternatives. Moreover, no systematic tools are currently available for sanitation planners to analyze and compare the lifecycle environmental impacts of alternative sanitation approaches, including the potential costs and benefits of resource recovery. This is particularly striking given rapidly growing unserved populations, the SDG target of universal sanitation access, diminishing resources, increasingly variable climate conditions and the potential for conventional sanitation infrastructure to be resource intensive, environmentally harmful and expensive to build and operate.

Life cycle assessment (LCA), a technique that studies the environmental aspects and potential impacts throughout a product's life, has been used in the wastewater sector since at least the mid-1990s to quantify and compare environmental impacts (ISO, 2006; Emmerson et al., 1995, Tillman et al., 1998, Dennison et al., 1998). The majority of these studies have focused on centralized, conventional wastewater treatment plants in high-income countries (Corominos et al., 2013). These studies consistently report substantial environmental impacts, particularly regarding energy use and global warming potential (GWP), of conventional wastewater treatment plants (Emmerson et al., 1995; Tarantini et al., 2001; Pillay et al., 2004; Friedrich et al., 2007; Machado et al., 2007, Ortiz et al., 2007; Kalbar et al., 2013). A handful of LCA studies have investigated the environmental impacts of passive, decentralized wastewater treatment infrastructure, although no studies conducted in SSA were found in the peer-reviewed literature (Dixon et al., 2003; Machado et al., 2007; Fuchs et al., 2011). Two LCA studies were found which investigate environmental impacts of dry, on-site sanitation (Flores et al., 2009; Friedrich et al., 2009). However, no studies were found that address the full lifecycle costs of non-sewered approaches in a low-income urban context. Life cycle cost (LCC) approaches have also been applied to sanitation infrastructure, however available studies are primarily focused on specific technologies (Fonseca et al., 2010). Related literature suggests that downsides to non-sewered approaches may exist. For example, there may be substantial environmental costs of transporting excreta and urine *via* use of fossil fuels rather than water (Tillman et al., 1998; Lundin et al., 2000; Flores et al., 2009). Other literature suggests decentralized approaches may incur additional financial costs associated with poor economies of scale (Whittington et al., 2008).

Given the considerable resource requirements of sanitation infrastructure, coupled with increasingly variable climatic conditions and resource scarcity, it is essential that water and energy requirements are integrated into planning for sanitation investments. At a global level, it is also critical that infrastructure decisions seek to minimize carbon emissions. It is likely that no single "best" approach to urban sanitation exists as costs and benefits of different strategies are strongly shaped by contextual socio-economic, physical and institutional factors. Recovery of resources from waste may help to alleviate environmental, financial and resource constraints. However, evidence-based research and analytical tools are necessary to quantify the possible advantages of recovery strategies.

Accordingly, the objectives of this dissertation are to (1) provide quantitative, empirical evidence on the costs and benefits of integrated sanitation and energy recovery strategies in low-income countries, (2) develop a lifecycle based framework to investigate the resource use, environmental impacts and financial costs and benefits of decentralized sewered and non-sewered sanitation approaches and (3) demonstrate use of this framework with primary and secondary data from operational sanitation systems in Zambia and India, thus also generating the first comprehensive analyses of decentralized sewered and non-sewered sanitation management in urban SA and SSA. The specific research questions examined in this dissertation towads achieve these objectives are:

1. To what extent does adoption of household anaerobic digesters impact (1) conventional fuel use, (2) GHG emissions and (3) financial expenditures on household cooking fuel? *(Chapter 2)*
2. What is the treatment performance of communal anaerobic digesters used for domestic wastewater management? *(Chapter 3)*
3. To what extent, and under what conditions, does biogas recovery from domestic wastewater offset GHG emissions and financial costs of sanitation service delivery? *(Chapter 3)*
4. How can the lifecycle resource use, carbon and financial costs and benefits of decentralized sewered and non-sewered sanitation approaches be quantified? *(Chapter 4)*
5. What are some of the key drivers and to what extent does energy recovery affect the lifecycle costs and benefits of decentralized sewered and non-sewered sanitation? *(Chapter 4)*

1.3 Overview of dissertation

This dissertation is organized as three studies, each of which is written as a manuscript. The remainder of the dissertation is organized as follows:

Chapter 2 – through collection and analysis of primary data from domestic, agricultural biogas digesters in Northern Tanzania, this study contributes empirical evidence on the environmental and economic impacts of a small-scale waste-to-energy and agricultural strategy in a low-income rural context.

Chapter 3 – this study contributes empirical evidence on the costs and benefits of an integrated wastewater treatment and energy recovery strategy in peri-urban Zambia. Specifically, this study investigates (1) the treatment and energy performance of communal ADs, (2) the extent and conditions under which biogas recovery may offset conventional energy use and greenhouse gas (GHG) emissions, and (3) the financial viability of biogas energy recovery from domestic wastewater under a range of scenarios.

Chapter 4 contributes to the knowledge and practice of urban sanitation planning by developing a lifecycle assessment (LCA) and lifecycle cost (LCC) based framework to evaluate the lifecycle water and energy use, GHG emissions and financial costs and benefits of sanitation infrastructure alternatives. Secondly, use of this framework is demonstrated with primary data from four operational sanitation systems in Zambia and India, thus also generating the first comprehensive analyses of the costs and benefits of non-sewered and sewered systems in urban SSA and SA.

Chapter 5 summarizes the contributions of this work, discusses implications for sanitation planning and identifies areas of future research.

2 Economic and environmental impacts of domestic bio-digesters: Evidence from Arusha, Tanzania

A modified version of this chapter has been published in Energy for Sustainable Development under the title "Economic and environmental impacts of domestic bio-digesters: Evidence from Arusha, Tanzania."[2] *Jennifer Davis is a co-author on the publication for her contribution to study design, data interpretation and manuscript revisions.*

2.1 Abstract

Despite substantial programmatic investment in domestic bio-digesters across sub-Saharan Africa in recent years, little empirical evidence has been published regarding the existence or magnitude of socioeconomic or environmental benefits accruing from bio-digester implementation. A cross-sectional study of 40 households in Arusha, Tanzania, suggests that bio-digester adoption has the potential to reduce fuel-wood use, energy-related expenditures, and time-costs of energy procurement; to lower CO_2e emissions; and to increase farm incomes. No significant differences in synthetic fertilizer use were observed between households with and without bio-digesters. Domestic bio-digester investments were found to have a positive net present value across a wide range of discount rates. Further, we estimate that domestic bio-digester implementation at the countrywide level in Tanzania could potentially access \$80–\$115 million annually in carbon emissions reduction (CER) financing through the Clean Development Mechanism.

[2] Laramee, J., & Davis, J. (2013). Economic and environmental impacts of domestic bio-digesters: Evidence from Arusha, Tanzania. Energy for Sustainable Development, 17(3), 296–304. http://doi.org/10.1016/j.esd.2013.02.001

2.2 Introduction

Domestic biogas digesters (or *bio-digesters*) decompose biodegradable waste, such as livestock manure, agricultural waste, and human excreta, through an anaerobic digestion process that allows for the recovery of energy (*biogas*) and nutrient (*bio-slurry*) resources (Mang and Li, 2010). Bio-digesters have a long history in China and India, where an estimated 40 million and 4.3 million, respectively, have been installed since the 1930s (Chen et al., 2010; REN21 Secretariat, 2011). In contrast, the potential for bio-digester use is in sub-Saharan Africa is relatively unrealized to date (Parawira, 2009). Introduction of bio-digesters in sub-Saharan Africa began in the 1950s, with the first large-scale implementation programs occurring in the 1980s (Gichohi and Wesenberg, 1990). Whereas the estimated potential (based on domestic cattle ownership and access to water)[3] for domestic bio-digester use in Africa has been estimated at 18.5 million (Heegde and Sonder, 2007), available data suggest that by 2005 only 7,000 bio-digesters had been installed in the region, of which 4,000 are estimated to be in Tanzania (Karakezi, 2009; Nhete and Kellner, 2007).[4] More recently, several programs have been initiated that aim to increase the number of bio-digester installations in sub-Saharan Africa (ABPP, 2012; TDBP, 2012).[5] For example, the Africa Biogas Partnership Programme (ABPP) aims to install 70,000 domestic bio-digesters in six sub-Saharan African countries over the 5-year period ending in 2013 (ABPP, 2012).

Bio-digester investment programs have been promoted as a means of advancing renewable energy use, agricultural productivity, and waste management (ABPP, 2012;

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[3] Domestic defined by Heegde and Sonder, 2007 as at least night stabled.

[4] Comparatively high numbers of bio-digesters installed in Tanzania can be attributed to efforts by the Centre for Agricultural Mechanisation and Rural Technology (CAMARTEC), which has been involved in the development, promotion and construction of biogas in the country.

[5] Substantial efforts to increase bio-digester installations worldwide are also ongoing: an estimated 22 million new bio-digester plants were installed in China between 2006 and 2011 (REN21 Secretariat, 2011). Major biogas programs are also underway in Vietnam, Nepal, and India, where a total of 105,000 bio-digesters were installed in 2010 (REN21 Secretariat, 2011).

TDBP, 2012). A large grey literature, produced largely by and for organizations implementing bio-digester projects, claims a variety of socioeconomic, environmental, and health benefits at the household, community, and global level (ABPP, 2012; Sasse et al., 1991; TDBP, 2012; Tumwesige et al., 2011). Such claimed benefits of bio-digesters include decreased use of conventional cooking fuels, which reduces the household's financial expenditures and time required for fuel collection, improves indoor air quality, and decreases greenhouse gas (GHG) emissions (Bond and Templeton, 2011; Gautam et al., 2009). Use of bio-slurry is also purported to increase agricultural productivity and decrease reliance on synthetic fertilizers (Werner et al., 1989). Benefits of domestic bio-digesters are said to accrue particularly to women and children, who are typically responsible for fuel collection, cooking, and small-holder agricultural activities in sub-Saharan Africa (Kes and Swaminathan, 2006; Saito, 1994; Van Nes and Nhete, 2007).

By contrast, the peer-reviewed literature includes only a handful of studies that evaluate the existence or magnitude of bio-digester benefits, particularly in sub-Saharan Africa (Mshandete and Parawira, 2009). Moreover, contradictory evidence exists among the few studies available. Regarding impacts on fuel use and expenditures, Xiaohua et al. (2005) found that households with bio-digesters in Eastern China used 60% less conventional energy, measured as kilograms of coal equivalent (kgce), compared to households without bio-digesters. Mwakaje (2008) reported that in Rungwe, southwest Tanzania, households with bio-digesters spent 96% less on energy related expenses realized compared to households without bio-digesters. By contrast, in a study conducted in North and Southwest China, Groenendaal and Gehua (2010) reported few or no benefits in terms of energy use, GHG emissions, energy financial expenditures, time savings, farm income or synthetic fertilizer use for households with bio-digesters as compared to those without.

Investment in bio-digesters has increased sharply in recent years despite the dearth of evidence regarding the extent to which these and other benefits are realized. This study seeks to contribute to this knowledge gap through collection and analysis of primary data regarding the extent to which households with bio-digesters (*versus* those without) realize benefits in terms of: (1) monetary savings; (2) time savings; and (3) increased farm income. In addition, the impact of domestic bio-digester adoption on conventional fuel consumption and greenhouse gas emissions equivalents is evaluated. The study makes use of data collected from households in the Arusha region of northern Tanzania.

2.3 Methods

2.3.1 Study site

The study was carried out in seven communities located between 3 to 24 kilometers northwest of Arusha town in northern Tanzania. The topography is dominated by Mt. Meru, with study site elevations ranging between 1400-1600 meters. Average annual precipitation is approximately 960 millimeters, with a bimodal rainfall pattern: the ‘long rains’ occur from mid-March to June and the ‘short rains’ from October to December. Temperatures in the region range from average lows of 17°C in July and August to average highs of 21°C in March (Ong’or and Long-cang, 2007).

House construction in the study region consists primarily of concrete block, earth brick, timber or wattle-and-daub. The majority of the rural population is engaged in semi-subsistence agriculture, with bananas, coffee, vegetables, beans, and maize commonly grown for both household consumption and income generation. Households typically own at least one head of high-grade dairy cattle, commonly stabled both day and night (termed “zero-grazing”), and practice integrated farming methods whereby manure is used for crop production (Kombe and Hoffman, 1990).

The Centre for Agricultural Mechanisation and Rural Technology (CAMARTEC) has administered two major domestic bio-digester programs in the Arusha region: first through the Biogas Extension Programme (BES) in the 1980s-90s and currently through the Tanzania Domestic Biogas Programme (TDBP) (ABPP, 2012; Kombe and Hoffman, 1990). Both CAMARTEC programs began their operations in Arusha.

Consequently, a high concentration of both new and old bio-digesters exists within the study region.

2.3.2 Research design

A cross-sectional research design was used to compare differences in energy consumption, energy expenditures, time savings, GHG emissions, farm income, and synthetic fertilizer use between households with and without bio-digesters. The study objective was thus to gain insights about the impacts of bio-digester use, rather than to determine the share of installed bio-digesters that succeed *versus* fail, which other studies have reported (Bensah and Brew-Hammond, 2010; Bond and Templeton, 2011; Chen et al., 2010; Van Nes and Nhete, 2007). As such, only functioning bio-digesters - defined as facilities with no gas leakages that are used regularly by the household - were included in the study. A total of 20 households that have adopted bio-digester technology (*adopters*) and 20 households without bio-digesters (*non-adopters*) were included in the study sample.

A purposive sampling approach, using information provided by the Tanzania Domestic Biogas Programme (TDBP) and associated key informants, was used to identify households with bio-digesters. Non-adopter households were identified based on information elicited from heads of adopter households, who were asked to identify a neighbor with similar socio-economic characteristics who did not have a bio-digester. The study is thus limited by a relatively small sample size, a non-randomly assigned treatment (adopter households are a self-selected group), and cross-sectional data. As selection of households was purposeful, rather than random, results must be viewed as illustrative rather than representative. A total of 22 adopter and 20 non-adopter households were asked to participate, of which two adopter households declined. Oral informed consent was obtained from either the male or female head of each participating household.

All bio-digester systems included in the study are located at single-family homes and are a modified Chinese fixed-dome type, constructed with locally burnt bricks and cement/lime plaster. The digester volumes range from $6m^3$ to $16m^3$ (Figure 1). Current (2011) cost of construction in Tanzania for such bio-digesters, including gas piping to the household, gas stove, and labor, ranges between approximately $750 ($6m^3$ volume) to $1,050 ($13m^3$ volume) (TDBP, 2011). Nine of the bio-digesters included in the study were built prior to 1995 and the remaining eleven were built after 2009. Nine of the adopter households surveyed had a pour-flush toilet connection to their bio-digester system. Households reported use of biogas primarily for domestic purposes, including cooking and boiling water for drinking and hygiene. Additionally, 45% of households reported using biogas to heat drinking water for cattle during the two-month cold season, and/or for small-scale entrepreneurial activities.

Figure 2.1: Bio-digester under construction: Arusha, Tanzania (Laramee, 2011)[6]

[6] Main biogas dome completed, manhole to expansion chamber (yet to be completed) in foreground, and piping connections to manure mixing chamber and toilet (yet to be completed) shown in background.

2.3.3 Data collection

Data collection took place over a seven-week period during the months of July and August, 2011. A total of 40 semi-structured interviews were conducted with adopter and non-adopter heads of households, of which 27 (67.5%) were with the female head; one (2.5%) was with the male head; and in 12 (30%) interviews both heads of household were present. The interviews were conducted in Kiswahili, with responses translated to and recorded in English at the time of interview. Detailed information was collected regarding the household's energy use, time and money costs of energy supplies, synthetic fertilizer use, and farm income.

2.3.3.1 Conventional fuel use

Daily fuel-wood (firewood and charcoal) use by all households was physically estimated by weighing household supplies at the time of interview using a spring-balance. Use of kerosene and liquefied petroleum gas (LPG) was self-reported by respondents.

2.3.3.2 Biogas use

Household biogas consumption was measured using an Elster AMCO Water G4 200 CFH Gas Meter (Elster Group; Essen, Germany). A HOBO 4-Channel Pulse Data Logger (Onset Computer Corporation; Bourne, MA) was used to record biogas use at 15-minute intervals. The gas meter and data logger operated for an average of 58.4 hours (SD=12.2 hours; range=45.5 - 72.3 hours) at 17 out of the 20 adopter households. The devices were positioned so as to avoid interference with the household's cooking and other biogas-using activities. The methane (CH_4) content of biogas was estimated indirectly by measuring the carbon dioxide (CO_2) content, using a CO_2-indicator (Brigon Messtechnik GmbH; Rodgau, Germany).[7] Four CO_2 tests

[7] Biogas consists of primarily CH_4, CO_2, and up to 2% other gases (Mang and Li, 2010) and thus, CH_4 content was estimated to make up the remaining percentage of gas after the average CO_2 content plus two percent for other trace gases was subtracted.

were performed at each of the 17 households (two tests at the time of gas meter installation and two tests immediately after the period of biogas metering).[8]

2.3.3.3 Energy expenditures

Monetary energy expenditures for non-adopters include the cost of purchasing fuel, as well as payments to hired labor for transportation and/or fuel preparation (e.g., chopping firewood). Energy expenditures for adopters include operation and maintenance costs of the bio-digester, as well as costs of supplementary fuel use. Operation of a bio-digester requires collection of manure, mixing of manure with sufficient liquid (water or urine), removal of hard materials (e.g. straw), and putting manure into the bio-digester (a process collectively referred to as *feeding* the bio-digester). Several households in the study reported employing a farm laborer whose responsibilities included caretaking of cattle and bio-digester feeding. Operational costs include payment for hired labor, pro-rated according to percentage of time spent on bio-digester operations. Cost of water inputs are also included, although study households generally reported that cattle urine collected was a sufficient amount of liquid for mixing, and that water inputs were minimal.

2.3.3.4 Time spent on energy procurement and manure management

Time spent on daily energy procurement was self-reported by households. For non-adopters, time cost includes fuel collection, chopping firewood, and fire preparation. For adopters, the time costs of procuring energy include time spent feeding the bio-digester and time spent procuring supplementary fuel sources (e.g., firewood). Respondents were also asked to identify the proportion of time spent by each individual (family member or hired laborer) responsible for various tasks related to energy procurement.

[8] Biogas and CO_2 measurements were not performed at three of the 20 adopter households because of time constraints.

Because manure management practices are affected by bio-digester use, households' self-reported time spent on manure management was also recorded. Typical manure management reported by non-adopters included daily application of manure to the farm (typically transporting manure with a bucket or wheelbarrow), composting prior to application, or storage of manure slurry in a pit. In contrast, adopters commonly had gravity irrigation channels connected to the outlet of the bio-digester, which automatically transported the digested bio-slurry to the farm. The time and effort required for cleaning of the cattle shed was assumed to remain constant regardless of bio-digester installation.

2.3.3.5 Agricultural productivity & synthetic fertilizer use

Four complications arise when analyzing agricultural productivity of sample households. First, households typically cultivate several crop varieties in a common plot. Farm income, rather than a physical measure of crop production, is thus used as a proxy to evaluate agricultural productivity across households. Second, households consume a substantial portion of the crops they cultivate, and thus analyzing only farm income is likely to provide an incomplete picture of the total agricultural productivity of the household. Therefore, farm income, purchases on agricultural items, as well as synthetic fertilizer costs are considered in combination. Third, to determine effects on agricultural productivity resulting specifically from use of bio-slurry, the income generated, inputs for crops, and agricultural purchases of crops grown on plots more than 1km from the location of the bio-digester were excluded from the study. Finally, adopters and non-adopters in the study have slightly different average household plot sizes (Table 1). Farm income and synthetic fertilizer use is thus calculated on a *per*-acre basis to enable direct comparison. In total, a measure of combined farm income is calculated as household farm income (*per* acre), less household agricultural food expenses, less synthetic fertilizer expenses (*per* acre).

2.3.3.6 Ethics

The study was reviewed by Stanford University Human Subjects Research (HSR) and waived as exempt from IRB review. Prior to conducting the interviews, all study candidates were briefed regarding the details of the study, given the opportunity to ask questions and receive answers, and asked for oral consent for study participation.

2.4 Findings

2.4.1 Household characteristics

At the household level, adopters and non-adopters are similar with respect to most measured socio-economic and demographic characteristics (Table 1). All respondents engage in small-scale agriculture and all own the land on which they live. One exception is household electricity supply, installed in the homes of 80% and 50% of adopters and non-adopters, respectively.

Table 2.1: Household characteristics, adopter *versus* non-adopter households

	Adopters (*n*=20)	**Non-adopters (*n*=20)**
Mean (SD) number of persons in household	6.1 (1.5)	5.9 (2.2)
Male head of household: mean (SD) years of education	9.0 (3.5)	8.9 (3.5)
Female head of household: mean (SD) years of education	8.5 (3.4)	8.2 (2.2)
Mean (SD) plot size (acres)	0.83 (0.50)	0.65 (0.58)
% of households with private water supply [a]	90%*	65%
% of households with electricity supply	80%*	50%
% of households with masonry home construction [b]	100%	90%

[a] Includes private tap in yard or household.
[b] Includes concrete block or burnt earth brick construction.
Test of means for adopters *versus* non-adopters: *$0.05 < p < 0.10$.

2.4.2 Energy use

Other than biogas, firewood was the primary cooking fuel used by both adopters and non-adopters (Table 2). Most households (75% of adopters and 80% of non-adopters) reported use of more than one type of cooking fuel, with use patterns affected by seasonal variation and fuel prices. Exclusive use of biogas for cooking was reported by 25% of adopters.

Mean firewood and kerosene usage was found to be significantly lower for adopters as compared to non-adopters (both $p < 0.005$): on average, sample adopter households use 93% (5,376kg) less firewood *per* year and 98% (48L) less kerosene *per* year as compared to non-adopters. Non-adopters also reported higher use of charcoal and LPG as compared to adopters, although usage is not significantly different (both $p > 0.20$).

2.4.3 Energy expenditures

Mean annual monetary expenditures on energy are significantly higher for non-adopters compared to adopters ($p < 0.001$), principally driven by the difference in fuel-wood consumption of the two groups (Table 2). Average maintenance costs are calculated from reported expenses incurred by adopters with bio-digesters that are two years or older.[9,10] Commonly reported maintenance requirements include replacement of the stove (or inside ring burner) due to deterioration over time, as well as removal of blockages resulting from a hardened top layer of bio-slurry formed within the bio-digester. In total, adopter households spent an average of $249 less *per* year on energy

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[9] Present value of past maintenance costs for older bio-digesters (installed at least 2 years prior to data collection) were calculated using Tanzanian Consumer Price Index (CPI) and 2011 USD exchange rate (World Bank, 2011). The present value of combined maintenance costs was then distributed annually over the lifespan of older bio-digesters. Alternative profiles of maintenance costs did not affect study conclusions.

[10] Households with bio-digesters built within two years prior to data collection did not report any maintenance issues or costs.

as compared to non-adopter households, not including the value of the household members' time.

Table 2.2: Annual fuel consumption and expenditure, adopter *versus* non-adopter households

		Fuel consumption		Fuel expenditure [b]	
		Adopter (n=20)	**Non-Adopter (n=20)**	**Adopter (n=20)**	**Non-Adopter (n=20)**
Firewood [a] (kg/year)	Mean	415	5,791***	$14.26	$188.23***
	(SD)	(640)	(3,134)	(45.65)	(170.51)
	Median	*88*	*5,617*	*0.00*	*143.78*
Charcoal (kg/year)	Mean	25	61	$14.05	$15.86
	(SD)	(73)	(106)	(40.39)	(28.10)
	Median	*0*	*0*	*0.00*	*0.00*
Kerosene (L/year)	Mean	1	49**	$0.76	$66.28**
	(SD)	(3)	(67)	(3.39)	(91.06)
	Median	*0*	*12*	*0.00*	*7.95*
LPG (kg/year)	Mean	0	17	$0.00	$21.21
	(SD)	(0)	(57)	(0.00)	(65.70)
	Median	*0*	*0*	*0.00*	*0.00*
Bio-digester operations	Mean			$10.89	
	(SD)	N/A	N/A	(18.86)	N/A
	Median			*0.92*	
Bio-digester maintenance	Mean			$2.81	
	(SD)	N/A	N/A	(2.20)	N/A
	Median			*2.97*	
Total	Mean			$42.77	$291.58***
	(SD)	N/A	N/A	(75.04)	(210.98)
	Median			*17.03*	*261.68*

[a] Figures reported are dry weight, estimated using a moisture-content of 13% (Simpson, 1998).
[b] $1 = 1,584 TZS (Oanda.com, 2011).
Test of means for adopters *versus* non-adopters: **$0.001 < p \leq 0.005$ ***$p \leq 0.001$.

2.4.4 Energy: time impacts

In total, adopter households were found to spend an average of 1.4 fewer person-hours *per* day on energy procurement and manure management activities as compared to non-adopters ($p < 0.001$) (Table 3). For 76% of non-adopter households, the female head of household is responsible for procuring energy; the male head of household is responsible in 4% of households (Figure 2). In contrast, females are responsible for energy procurement in 37% of adopter households ($p < 0.005$), and males in 16% of households ($p=0.15$).

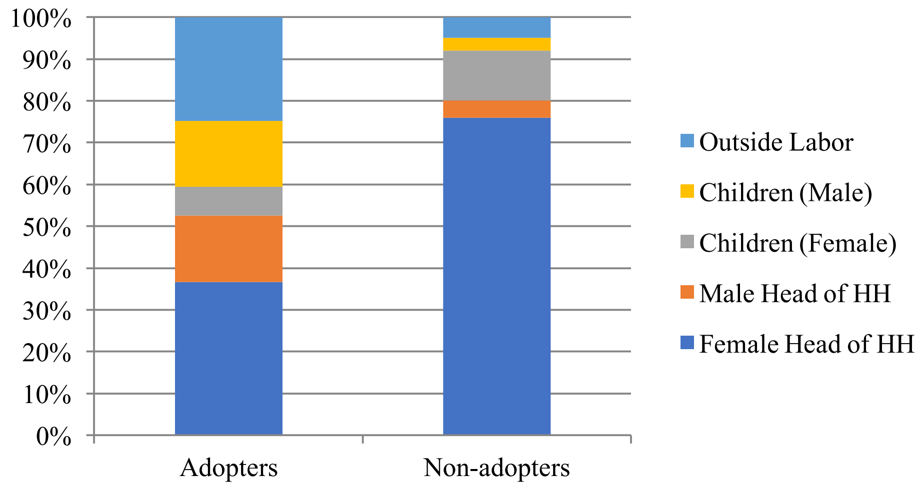

Figure 2.2: Allocation of time spent collecting and preparing fuel, adopter *versus* non-adopter households

Table 2.3: Daily time costs (minutes/day/household) for energy procurement and manure management: adopters *versus* non-adopter households

Activity		**Adopters (n=20)**	**Non-adopters (n=20)**
Collecting conventional fuel	Mean (SD)	2 (6)	56 (51)***
	Median	*0*	*45*
Preparing conventional fuel	Mean (SD)	4 (6)	28 (21) ***
	Median	*2*	*22*
Feeding bio-digester	Mean (SD)	27 (19)	N/A
	Median	*26*	
Manure management	Mean (SD)	26 (24)	61 (42) **
	Median	*19*	*60*
Total	Mean (SD)	59 (29)	145 (57) ***
	Median	*51*	*145*

Test of means for adopters *versus* non-adopters: energy procurement **$0.001 < p < 0.005$, ***$p < 0.001$.

2.4.5 Greenhouse gas emissions & biogas use

The combustion of wood and fossil fuels results in the emission of carbon dioxide (CO_2) and methane (CH_4) as significant GHGs. However, gathering of firewood by individuals in rural settings is generally considered to have minimal impact on deforestation, and thus net zero carbon emissions, as dead wood or branches are typically gathered rather than felling entire trees (UNDP/World Bank, 2001). Firewood harvested for commercial purposes, however, typically requires harvesting entire trees and thus is generally considered to have a greater impact on deforestation and consequently a net positive effect on carbon emissions (UNDP/World Bank, 2001). Similarly, charcoal production typically results in felling entire trees, and thus can be considered to have a net positive impact on carbon emissions (Chidumayo and Gumbo, 2012). Thus, for the purposes of tabulating the aggregate impact that cooking fuel consumption has on GHG emissions for adopters *versus* non-adopters, the mean fuel consumption for each set of households was adjusted considering gathered firewood as renewable (zero emissions) and commercial firewood and charcoal as non-renewable (contributing to emissions). Overall, approximately 50% of adopter household's firewood source was commercial (annual mean = 207kg/yr, SD = 644kg/yr), while 60% of non-adopter household's firewood source was commercial (annual mean = 3,473kg/yr, SD = 3798kg/yr, median = 1988kg/yr). Mean fuel consumption was then converted to CO_2-equivalent (CO_2e) emissions using emissions *per* unit mass of fuel, and respective global warming potentials (GWP) (Table 4). On average, adopters generate 5,203 kg less CO_2e *per* year *per* household as compared to non-adopters, a statistically significant difference ($p<0.001$).[11]

[11] The results of this analysis depend on the share of renewable *versus* non-renewable firewood used by households. Assuming all (100%) firewood use is renewable (i.e. zero emissions), adopter households on average generate 386 kg less CO_2e per year per household as compared to non-adopters ($p=0.03$). However, if all firewood is considered non-renewable, adopter households generate on average 8,317kg/yr less as compared to non-adopters ($p<0.001$).

Table 2.4: Annual CO_2e emissions: adopter *versus* non-adopter households

	Emissions *per* unit mass of fuel combusted (g/kg)		**Annual CO_2e emissions (kg/year)**	
	CO_2	**CH_4**	**Adopters (n=20) Mean (SD) *Median***	**Non-Adopters (n=20) Mean (SD) *Median***
Global Warming Potential [a]	1	21[b]	-	-
Firewood [c]	1370	5	306 (950) *0*	5,122 (5,602)*** *2,932*
Charcoal (use) [d]	2400	6	78 (229) *0*	189 (331) *0*
Charcoal (production) [d]	2085	29	79 (233) *0*	192 (336) *0*
Kerosene [e]	3050	0.9	2 (9) *0*	155 (214)** *38*
LPG [e]	3190	0.01	0 (0) *0*	54 (186) *0*
TOTAL			437 (1,092) *0*	5,640 (5,694)*** *3,918*

[a] UNFCC (1995)
[b] The 100-yr GWP of methane has been revised to 25 *per* errata to the IPCC Fourth Assessment Report (IPCC, 2012)
[c] Bailis et al. (2003)
[d] Kammen and Lew (2005)
[e] Smith (1994)
Test of means for adopters *versus* non-adopters: **$0.001 \leq p \leq 0.005$, ***$p < 0.001$.

Adopters were found to use an average of 1.85 m^3 of biogas *per* day *per* household. The average CH_4 content of biogas among adopter households was measured at 64% (SD=2.2%). When combusted, CH_4 is converted into the less potent GHG CO_2, thus reducing the overall global warming potential from 21 to 1 (UNFCC, 1995). Thus, an average of 5,825 kg CO_2e emissions are captured *per* year *per* sample adopter household through the combustion of CH_4 for cooking (Table 5). However, the degree to which this figure represents a reduction in CO_2e emissions as compared to non-adopter households is dependent on manure management practices, as well as the extent to which biogas produced is utilized by adopter households.

Table 2.5: Biogas consumption and CH_4 content, adopter households (*n=17*)

Biogas consumed *per* day *per* household (m^3/day)	Mean	1.85
	(SD)	(0.48)
	Median	*1.92*
Biogas produced *per* day *per* cow (m^3/day)	Mean	0.77
	(SD)	(0.24)
	Median	*0.68*
CH_4 content of biogas	Mean	64%
	(SD)	(2%)
	Median	*63%*
CO_2e captured *per* year *per* household (kg/year) [a]	Mean	5,825
	(SD)	(1,616)
	Median	*6,001*

[a] Density of CH_4 = 0.67kg/m^3 (assuming average temperature of 17.5°C and 1 atm). CO_2e (kg/year) = 1.85m^3/day x 0.64 x 365 days *per* year x 0.76kg/m^3 x GWP 20 = 5,825 kg CO_2e/year.

2.4.6 Farm Income

The combined average annual farm income for adopter households was found to be \$384 *per* acre (SD=\$674, median=\$342 *per* acre), which is significantly higher than the income for non-adopters of -\$258 (SD=\$446, median=-\$320 *per* acre) ($p<0.005$) (Figure 3). Adopters were also analyzed in two groups based on the length of time since bio-digester installation (those with systems built prior to 1995 and the remaining with systems built after 2009), based on the assumption that changes in agricultural productivity resulting from the availability of bio-slurry may take longer than a few years to accrue. Contrary to expectations, adopters with newer systems are found to have a slightly higher mean annual farm income (mean=\$421, SD=\$655, median=\$354 *per* acre) as compared to those with older systems (mean=\$340, SD=\$734, median=\$329 *per* acre), although this difference is not significant ($p=0.80$).

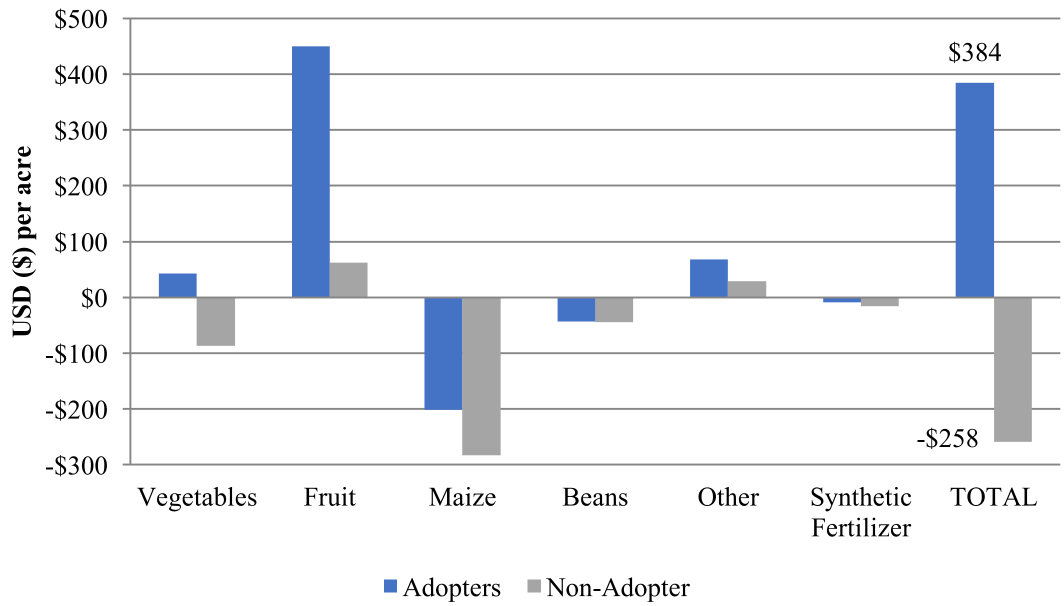

Figure 2.3: Combined mean annual farm income: adopter *versus* non-adopter

Adopters were also asked to comment anecdotally on the advantages and disadvantages of using bio-slurry for agriculture. Nearly all (95%) sample adopters considered bio-slurry to be superior both to fresh cow manure and to synthetic fertilizers. Respondents based their preference for bio-slurry on the belief that it maintains soil fertility; can be applied directly to crops without prior composting; includes a high share of water which is valuable for irrigation; improves soil texture; and improves the soil's ability to retain water. Similar benefits have been documented in the literature (Sasse et al., 1991; Werner et al., 1989).

2.4.7 Synthetic fertilizer use

Self-reported mean annual synthetic fertilizer use was 15kg *per* acre (SD=38kg) for adopter households and 23kg *per* acre (SD=51kg) for non-adopters, a difference that is not statistically significant (p=0.59). In general, reported use of synthetic fertilizers at home plots was low across all sample households: only 30% of adopters and 15% of non-adopters reported use of synthetic fertilizers at their household plot, while others reported using exclusively organic fertilizer (cow manure for non-adopters or bio-slurry for adopters). Synthetic fertilizer was more commonly used at secondary plots located at some distance away from the household, where bulk or liquid manure could not be effectively transported. Plots located more than 1km from the sample homes were not included in the analysis, however, in an effort to identify impacts related to slurry use.

2.4.8 Toilet connected systems

Eleven of the 20 adopter households had not connected their toilet to their bio-digester; among these, 64% cited the prohibitive cost of building a new toilet. The remaining respondents reported reluctance to use bio-slurry from a digester connected to a household toilet because of stigma or health concerns. Use of bio-slurry did not appear to be substantially different, however, between households with toilet-connected systems and those whose bio-digesters are operated with livestock and agricultural wastes only. One toilet-connected household reported non-use of bio-slurry on vegetable cultivation, but did report use of slurry for banana and coffee cultivation.

2.4.9 Financial net present value (NPV) and payback period

An NPV analysis of the decision to invest in a bio-digester was undertaken that considers the costs of digester construction, expenditure on wood and fossil fuels, and time costs of energy procurement and manure management (Table 6). Other potential benefits of bio-digester installation, such as increased farm income, health and, nutrition improvements, were not included in the analysis.

Assuming a minimum unskilled labor rate of USD $41 *per* month for a work week of 44 – 48 hours (US Dept. of State, 2008), and conservatively assuming 50% of this rate for women (the sample household members whose time is typically saved by bio-digester implementation), the average annual value of time saved is approximately $54 *per* year.[12] Thus, the combined annual savings (energy expenditures and value of time saved) is $303 for adopters in the sample. Assuming these energy and time-savings are realized every year, the simple payback period for a domestic size bio-digester is estimated to be 2.5 to 3.5 years.[13]

Discount rates of 15% and 35%, based on the current domestic loan interest rates charged by the Tanzania Central Bank and typical micro-finance lenders, respectively, were applied in the NPV analysis (Indexmundi, 2012; CGAP, 2008). These values were presumed to bound the range of available in-country discount rates for sample homeowners. An average lifespan of 15 years[14] and an inflation rate of 7.8% have been assumed to estimate inflation rates of future costs and benefits (World Bank, 2010). The internal rate of return (IRR) was found to be 50% and 38% for 6m^3 and 13m^3 volume bio-digesters, respectively. In each discount rate scenario, the NPV of both 6m^3 and 13m^3 bio-digesters was found to be positive at these high, but realistic, discount rates.[15]

Table 2.6: NPV analysis of bio-digester installation, assuming 15% or 35% discount rate, 15-year lifespan [a]

Digester size	**15%[b] discount rate**		**35%[c] discount rate**		**IRR[d]**
	NPV	Payback period (yr)	NPV	Payback period (yr)	
6 m^3	$2,046	2.9	$392	4.6	50%
13 m^3	$1,746	4.2	$92	9.8	38%

[a] TDBP subsidy not included in NPV and IRR calculations.
[b] Tanzania central bank discount rate (Indexmundi, 2012).
[c] Average micro-finance lending rate (CGAP, 2008).
[d] Internal rate of return.

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[12] This approach was found to be conservative by Whittington et al. (1990).
[13] A $200 subsidy is currently provided by the Tanzania Domestic Biogas Programme (2011) for bio-digester construction. Taking this into account, the simple payback period reduces to 1.8 to 2.8 years.
[14] The assumed 15-year lifespan of a bio-digester is thought to be conservative. Nine of the 20 bio-digester systems included in the study were built more than 20 years prior to interview, including 3 bio-digesters built 27 years prior.
[15] Although the larger size digester has a greater gas storage capacity, the annual financial benefit for each size digester was assumed to be equivalent as there was not a significant difference in annual energy consumption or savings found between different size digesters.

These favorable financial results for individual, functional bio-digesters do not necessarily translate into benefits at the programmatic level, given the risk of operational failure or abandonment of some digester systems. Van Ness and Nhete (2007), for example, estimate a 40% operational success rate for bio-digesters in sub-Saharan Africa. Similarly, Bond and Templeton (2011) report the proportion of functional domestic bio-digesters at 50% or less in developing countries excluding China and India. If these rates are typical for bio-digester programs in the study region, the net benefits presented above would need to be modified accordingly. The existence of functioning bio-digesters in the study sample that were constructed up to 27 years prior to the survey, however, demonstrates the potentially long lifespan of bio-digesters with sound construction, and with educated and motivated users.

Additionally, investigations in other locations could help elucidate the extent to which experience with household digesters in Arusha mirrors that in other regions. In particular, characteristics that would likely affect the beneficial economic impacts of bio-digester implementation are lack of cattle ownership, non-zero grazed cattle, or poor access to water. Studies in other regions with less optimal conditions can help to assess the generality of these findings or to refine them contingent on these variables, helping to target biogas programs to the regions where their net benefits are greatest.

2.5 Discussion and conclusions

Findings from rural Tanzanian households with and without bio-digesters suggest that, in this setting, digesters can reduce fuel-wood use, money and time costs of fuel procurement, as well as GHG emissions. By contrast, data are inconclusive regarding any systematic difference between adopter and non-adopter households in terms of agricultural productivity, and no evidence of impact on synthetic fertilizer use was observed. The fuel-wood savings is particularly noteworthy for Tanzania, where over 95% of the population is reported to rely solid fuel-wood to meet their basic energy needs (UNDP/WHO, 2009).

Among sample households, the magnitude of energy savings generally accords with that found by Mwakaje (2008) in Southwest Tanzania, but is higher than that found by both Xiaohua et al. (2007) and Groenendaal and Gehua (2010) in Eastern and Western China, respectively. The finding of comparatively higher energy savings in Africa may be due to differences in the type and amount of livestock manure available, availability of fuel-wood, and/or climatic conditions. For example, sample households in both Tanzania studies owned high-grade dairy cattle, whereas rural households with bio-digesters in China, commonly use pig manure as a feedstock. On average, dairy cattle produce 13 times more biogas *per* head than pigs, although this number varies widely based on size of livestock and type of feed used (Werner et al., 1989). Thus, a typical sample household in the China studies likely generates less biogas than a typical study household in Tanzania, and may require additional supplementary fuel sources.

Differences in the magnitude of fuel-wood savings found between regions may also be associated with the relative availability of such resources. Groenendaal and Gehua (2010) speculate that insignificant differences in fuel-wood consumption found between Chinese households with and without bio-digesters may be the result of ample supply of firewood and agricultural stalk residues in the study area. In contrast, Mwakaje (2008) lists shortage of firewood as a concern in the Rungwe district, Tanzania. Similarly, in the present study based in Arusha, 40% of sample adopter households cited increasing unreliability and scarcity of fuel-wood as a motivation for investing in a bio-digester.

Climatic conditions may also play a role in the magnitude of fuel-wood savings associated with bio-digester use. In general, the optimal temperature range for microbial digestion by methanogenic bacteria is 8°C to 65°C, with highest production occurring at approximately mid-range and minimal production below 8°C (Mang and Li, 2010). Xiaohua et al. (2005) note that biogas is produced at one of their field sites in China for only 9-10 months of the year, with no production during cold winter months. Temperatures in the Arusha, Tanzania region are high enough for biogas production to occur year-round, although sample households noted a perceived decrease in biogas production during the cold season (during which this study was undertaken).

The average fuel expenditure savings for adopter households in this study amounts to nearly $250 *per* annum, equivalent to 20% of the estimated annual income for an unskilled laborer in Tanzania (US Dept. of State, 2008). Energy prices in Tanzania have historically been volatile but have generally outpaced overall consumer inflation (Ng'wanakilala, 2012). Energy price increases, along with increasing fuel-wood demand as noted above, suggest that the savings realized through energy expenditure reductions can reasonably be expected to increase in coming decades.

Notably, the 1.4-hour average daily time savings observed for sample adopter households, resulting from reduced effort for energy procurement and manure management, accrues primarily to women. Prior research has demonstrated that the responsibility for collecting firewood in sub-Saharan Africa falls disproportionately on women (Kes and Swaminathan, 2006). In contrast, animal husbandry activities are generally male dominated (Charmes, 2006), which may explain the observed shift in energy procurement roles between genders. Indeed, for non-adopters in the study, the female head of household was responsible for energy procurement more than twice as often as female heads in adopter households. Thus, not only were adopters found to save time overall, but the time spent in adopter families on energy procurement was more evenly distributed across genders.

Adopter households as compared to non-adopters in the sample release on average approximately 5.2 fewer tonnes of non-renewable CO_2e emissions *per* year *per* household, resulting primarily from fuel-wood savings. Adopter households additionally capture on average 5.8 tonnes of CO_2e emissions *per* year *per* household through the collection and use of CH_4. However, because the percentage of CH_4 produced during manure decomposition varies with the extent of anaerobic (*versus* aerobic) conditions, the degree to which reductions in CH_4 emissions by adopter households contribute to overall reductions in CO_2e emissions is dependent on the alternative manure management practices of non-adopter households. Bio-digesters are designed to create ideal anaerobic conditions for maximum CH_4 production, whereas the manure management practices of typical non-adopters are likely to result in CH_4 production that is between 0.5% (for households applying manure directly to their fields on a daily basis) to 39% (for households storing manure in a slurry pit) of this maximum value (Safley et al., 1992; Steed and Hashimoto, 1994; IPCC, 2006).[16] Thus, a more realistic estimate of annual CO_2e offsets resulting from CH_4 emission reductions may be negligible when the alternative practice is daily spread of manure to cropland, or may be up to 2.3 tonnes CO_2e *per* household when the alternative practice is to store manure in a slurry pit. Moreover, a household with a bio-digester may release CH_4 into the environment if it does not use all the biogas it produces, or if leakages exist in the digester system. In sum, the actual offsets in CO_2e emissions for adopter *versus* non-adopter households will depend on contextual factors such as manure management practices and biogas utilization by households.

Reducing fuel-wood use and associated emissions can have important impacts on human health. Each year nearly 450,000 premature deaths in sub-Saharan Africa are attributed to respiratory illness caused by indoor air pollution from cooking, heating, and lighting with solid fuel-wood (WHO/Rehfuess, 2006). Fuel-wood combustion is characterized by low efficiency and high emission rates (Bailis, 2003; Smith, 1994). In contrast, biogas combustion is highly efficient and produces significantly fewer harmful emissions (Smith et al., 2000). Whereas the scope of the study did not include air quality testing, self-reported information on respiratory symptoms was collected. Three quarters of respondents said they felt their respiratory health and/or eye health had improved as a result of shifting from fuel-wood or kerosene to biogas for cooking. Large-scale transitioning from fuel-wood to clean burning technologies

[16] These figures, however, are highly variable and dependent on temperature, duration of storage, rainfall, moisture content and other aspects of treatment. Thus the IPCC specifies an uncertainty range of +/- 30% (IPCC, 2006).

such as biogas may thus offer the potential of substantial health impacts in sub-Saharan Africa.

Sample adopter households on average reported a significantly higher farm income as compared to non-adopters. Contrary to the expectation that adopters with older bio-digesters would have higher farm incomes as compared to adopters with newer systems, however, no significant difference was found between these two groups. In fact, adopters with new systems had slightly higher reported farm incomes as compared to those with older systems. It is thus unclear whether higher farm incomes are in part a result of bio-digester implementation, or the result of other factors not fully captured by the study *(e.g.*, cattle ownership).

Cultural resistance to the use of resources recovered from human excreta has been noted in the literature as a possible barrier to successful implementation of waste-reuse technologies, particularly in sub-Saharan Africa (Chaggu et al. 2002; Duncker et al., 2007; Mariwah and Drangert, 2011). Evidence from this study suggests that some opposition to reuse of resources derived from human excreta exists, but may be overcome with experience. Of the nine households with toilet-connected systems, only one reported opposition to bio-slurry use for vegetable production. Notably, this household was also the only adopter with a toilet connection that had installed the digester within two years of interview. Moreover, the actual use of bio-slurry for households with toilet connections was not significantly different from that of households without toilet-connected systems. No households reported cultural opposition to biogas use for toilet-connected systems. Nor did any household report adverse health effects that they attributed to slurry use.[17]

The Millennium Development Goals (MDGs) statistics on access to sanitation have remained dismally stagnant in sub-Saharan Africa, particularly in rural areas. From 1990 to 2010, the percentage of the region's rural population with access to improved

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[17] Although the study did not include an analysis of pathogen inactivation through anaerobic digestion, others have evaluated these impacts (Mang and Li, 2010; Remais et al., 2009).

sanitation only increased from 19 – 23% (WHO/UNICEF JMP, 2012). In Tanzania, these numbers are even worse: by 2010, only 7% of the rural population had access to improved sanitation, up 1% since 1990 (WHO/UNICEF JMP, 2012). Willingness to pay for sanitation, particularly in rural areas of sub-Saharan Africa, is notoriously low (Whittington et al., 1998). Domestic bio-digesters offer energy, financial, and time-savings opportunities that other types of sanitation facilities do not. As such, it may be that incorporating renewable biogas energy into sanitation programs may incentivize households to invest in sanitation to a greater extent.

2.5.1 Limitations

As noted above, the study is limited by a small sample size and cross-sectional design. Although efforts were made to match adopter to non-adopter households with similar socio-economic characteristics, systematic differences between groups likely exist, in particular the self-selection of adopters according to their decision to invest in bio-digester technology. Further research would benefit from a larger sample size, the random assignment of bio-digester technology among study households, and the collection of longitudinal data.

2.5.2 Implications

Heegde and Sonder (2007) estimated the technical potential for domestic bio-digester implementation in Africa based on domestic cattle ownership and access to water. Using this methodology and latest Tanzania-specific data from the Food and Agricultural Organization (FAOSTAT, 2012) and the Joint Monitoring Program (WHO/UNICEF JMP, 2012), the technical potential for domestic bio-digesters in Tanzania is estimated to be 1.8 million. Based on typical household benefits found in the present study, if this potential were to be realized, annual benefits at the national level could be substantial: 9.8 million tonnes of fuel-wood saved, 9.4 to 13.5 million tonnes CO_2e emissions avoided, and 943 million person-hours saved, primarily for women. Such projections must be viewed as an upper bound, however, given the purportedly high rates of operational failure in many bio-digester programs (Bond and Templeton, 2011; Van Nes and Nhete, 2007).

Given the sizeable GHG emission offsets that widespread adoption of bio-digester systems in Tanzania could affect, accessing carbon emissions reduction (CER) financing through the Clean Development Mechanism (CDM) is worth consideration. Historically, CER financing has primarily been accessed by large, stand-alone projects due to the program's high transaction costs. Development of the Programme of Activities (PoA) modality, however, creates new opportunities for smaller and more decentralized investments (such as domestic bio-digesters) to enjoy financing, by spreading the transaction costs over a portfolio of projects that may be geographically and temporally dispersed (UNEP, 2009).[18] Indeed, CER financing has been approved for at least nine domestic biogas programs, primarily since 2011, in China, India, and Nepal, with at least 22 more programs currently in the validation process (ADATS, 2005; SEDS, 2012; UNFCC, 2012a).

CER financing may thus be a viable option for subsidizing the costs of domestic bio-digesters in Tanzania. The weighted average CER price for the post-2012 market was estimated in 2011 at US$11.5 (€8.3) *per* tonne CO_2e (World Bank, 2012).[19] Assuming emission offsets in the range of 5.2 to 7.5 tonnes CO_2e/year *per* household, along with investment programs that realize three quarters of Tanzania's digester

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[18] While still in the early stages of development, the PoA modality has the potential to substantially up-scale the number of CDM projects and improve regional access. For instance, by 2012, Africa based projects accounted for 28% of PoAs in the CDM cycle as compared to fewer than 3% in the stand-alone project-based CDM (World Bank, 2012).

[19] The carbon market has been volatile in recent years amidst a worldwide economic downturn compounded by a long-term oversupply of carbon allowances and uncertainty regarding the end of the first commitment period of the Kyoto Protocol (World Bank, 2012). CER prices dropped from an average high of over $16 per tonne CO_2e in 2008, to an average of $10.9 (€7.9) per tonne in 2011 (World Bank, 2012), and hit a low of 31 cents per tonne by the end of 2012 (Allan and Kruppa, 2012). Although, the World Bank projects that CER pricing for the post-2012 market will rebound back to at least 2011 average prices (World Bank, 2012), CER pricing has remained stagnant at all time lows despite the adoption of an amendment at the Doha 2012 UN Climate Change Conference extending the commitment period of the Kyoto Protocol an additional eight years to 2020 (UNFCC, 2012b).

technical potential, CER financing could be in the range of \$80 to \$115 million annually. Such financing could accelerate investment in domestic bio-digesters, potentially generating significant socioeconomic and environmental benefits for a larger share of Tanzania's population.

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3 Energy, carbon and financial costs and benefits of biogas recovery from communal anaerobic digesters: Evidence from peri-urban Zambia

This chapter is being prepared as a manuscript. Sebastien Tilmans will be co-author for contributions in data analysis, interpretation and manuscript revisions. Jennifer Davis will be co-author for contributions to study design, data interpretation and manuscript revisions.

3.1 Abstract

Communal anaerobic digesters (ADs) have been promoted as a waste-to-energy strategy that can improve sanitation service sustainability while providing clean energy. However, little empirical evidence is available regarding the performance of such systems under field conditions. This study assesses the wastewater treatment efficiency, energy production, greenhouse gas (GHG) emissions, and financial costs and benefits of communal ADs used for domestic wastewater treatment in Zambia. Primary data were collected over a 6-month period on the technical performance of 15 ADs. In-person interviews were also conducted with heads of 120 households regarding their use of biogas and conventional fuels. Chemical oxygen demand (COD) removal ranged from 61% to 83% and *per*-inhabitant biogas production ranged from 3L to 20L *per* day. Across all sites, 82% of biogas produced was recovered and used. Gas connected households used 56% less charcoal, 58% less electricity and 29% less total cooking energy compared to households without biogas supply (all $p \leq 0.01$). Households using ADs were estimated to have mean annual per-inhabitant GHG emissions ranging from 45 to 141 kg carbon dioxide equivalents (CO_2e) lower than similar households using conventional septic tanks. The net present value of the incremental investment needed to capture and distribute biogas from ADs ranged from -\$16 to \$47 *per* inhabitant, depending on biogas production rates, conventional cooking fuel used, biogas tariff and carbon pricing. Within these scenarios, a minimum carbon price between US\$9 and \$28 *per* tCO_2e is required to attract investment in emissions reduction projects such as ADs.

3.2 Introduction

Energy recovery from domestic wastewater has received increased attention in recent years as a strategy to offset operational costs of sanitation services, provide alternative energy resources and reduce environmental impacts of wastewater treatment (Guest et al., 2009; McCarty et al., 2011; Cornejo et al., 2013). This changing paradigm of waste as a resource rather than a costly problem creates opportunities for simultaneously addressing sanitation and energy challenges with a single approach. Both challenges are particularly pervasive in sub-Saharan Africa (SSA) where only 30% of the population has access to improved sanitation and more than 80% use fuelwood as their primary energy source (WHO/UNICEF, 2015; Legros et al., 2009). Moreover, limited public financing for initial or recurrent costs of sanitation infrastructure, coupled with low willingness to pay for wastewater treatment, contribute to poor sanitation services in many low-income countries (Whittington et al., 2000).

Anaerobic digestion technologies of varying complexity, arrangement and scale have been used throughout the world to treat organic wastes and generate energy. As organic waste biologically decomposes under anaerobic conditions it produces biogas, composed primarily of methane (CH_4), carbon dioxide (CO_2) and nitrogen (N_2) (Rittman and McCarty, 2001). The CH_4 content of biogas can be recovered and used as an energy source. Large-scale anaerobic digesters (AD) are commonly used at centralized wastewater treatment plants in industrialized countries to treat bio-solids produced by municipal wastewater treatment, with recovered CH_4 generally converted to electricity (Rittman and McCarty, 2001). Various centralized anaerobic treatment technologies have also been used for the direct treatment of dilute, domestic wastewater in middle-income countries in tropical climates, particularly in Latin America (McCarty et al., 2011).

Household-scale ADs have also been used in rural areas of developing countries for domestic energy production, particularly in China and India where 43 million and 4.75 million, respectively, installations are estimated (Bond and Templeton, 2011; REN21, 2016). In such installations, livestock manure is generally utilized as the main organic waste. Household-scale ADs operate based on similar anaerobic processes as large-scale systems, but are comparatively less complex, generally requiring no external energy inputs and limited monitoring of biological processes. Household facilities are also generally designed so that biogas can be used directly rather than converting it to electricity. Performance, costs and benefits of household ADs in a number of developing countries have been documented in the literature (Chen et al., 2010; Van Groenendaal and Gehua, 2010; Laramee and Davis, 2013).

Comparatively fewer communal ADs have been implemented for the treatment of domestic wastewater at the neighborhood or community level (e.g. serving approximately 10 – 100 households). Correspondingly little information is available characterizing treatment performance, energy production potential or financial viability of ADs at this scale (Reynaud, 2014; Tilmans et al., 2014). Through collection and analysis of primary data from communal ADs in Zambia, this study seeks to contribute empirical evidence on the costs and benefits of such infrastructure. In particular, the study investigates (1) the technical performance of communal ADs in terms of treatment efficiency and biogas energy production, (2) the extent to and conditions under which biogas recovery may offset conventional energy use and greenhouse gas (GHG) emissions and (3) the financial viability of biogas energy recovery from domestic wastewater under a range of scenarios.

3.3 Methods

3.3.1 Study sites

The study was carried out in three low- to middle-income peri-urban communities in the southern African country of Zambia. The three communities—referred to hereafter as Site A, B and C—are located within the cities of Solwezi (Northwestern Province), Ndola (Copperbelt Province) and Livingstone (Southern Province), respectively. Primary data collection took place over a 6-month period from March–August 2015, spanning the warm/wet season (mean highs: 25-30°C) to the cool/dry season (mean lows: 5-7°C) (World Bank, 2012).

Recent sanitation upgrade projects were implemented in each of the three study sites during the period 2008–2012. The projects included installation of new household flush toilets connected to small-bore sewer systems with integrated communal ADs as primary wastewater treatment units. In total, the study sites include 15 ADs, with each AD receiving wastewater from 9 to 67 households (Figure 3.1 and Table 1). At Site C, one AD (denoted C-PT) also receives waste from a public toilet. No other organic waste is added to the ADs. Domestic wastewater alone will provide only a fraction of a typical household's cooking energy requirements (Mang and Li, 2010). Thus, biogas is recovered and piped to, on average, 6% of households with toilet connections (1 to 4 households from each AD). Organic solids settle within the AD and undergo anaerobic digestion, while liquid-only effluent continues through the sewer network to secondary treatment systems. All study site ADs are fixed dome digesters constructed with burnt bricks and cement plaster. Each has a nominal reactor volume ranging from $16m^3$ to $36m^3$. The sanitation systems have been owned and operated by local water and sewerage utilities since 2012.

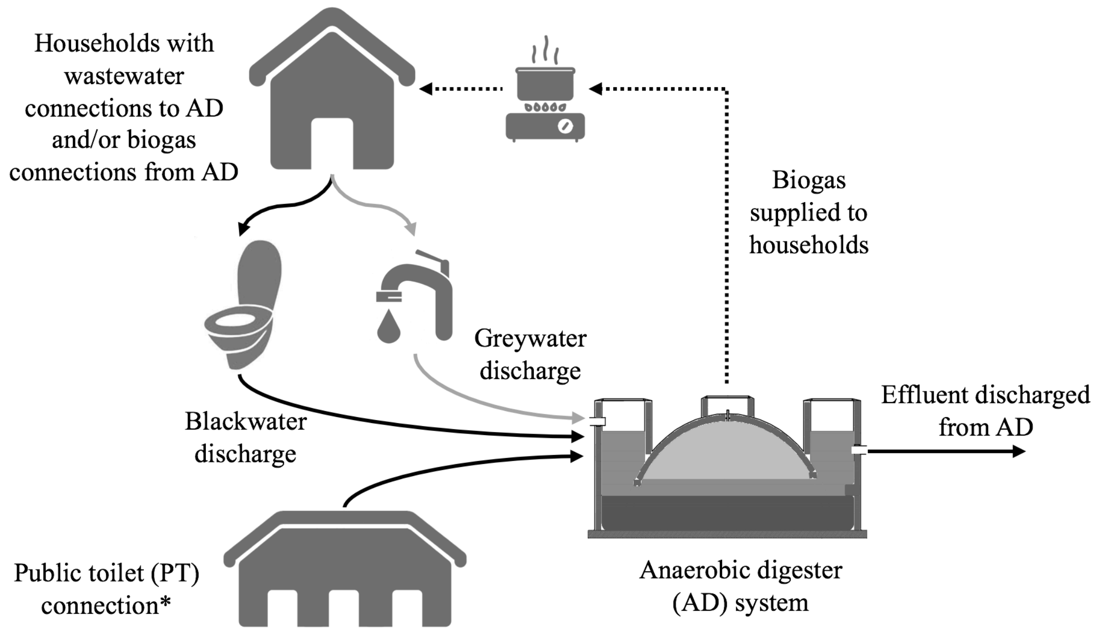

Figure 3.1: Wastewater flow and biogas supply diagram (*Public toilet connection at site C-PT only) (Images: SimGas, 2016; BORDA, 2016)

Table 3.1: Study site population, water supply, wastewater discharge and biogas supply characteristics

Parameter	Site A	Site B	Site C	C-PT
Number of anaerobic digester systems *per* study site	4	2	8	1
Number of households (*inhabitants*) with wastewater connections *per* study site	78 (*394*)	91 (*438*)	247 (*1548*)	15 (*77*) [a]
Number of households (*inhabitants*) obtaining biogas supply *per* study site	10 (*41*)	5 (*28*)	9 (*38*)	1 (*6*)
Household water supply infrastructure	In-house piped	Shared (public) water tap	Private yard tap	Private yard tap [a]
Type of toilet at household	Cistern-flush	Pour-flush	Pour-flush	Pour-flush
Wastewater type discharged into AD [b]	Greywater and blackwater	Blackwater	Blackwater	Blackwater

[a] Number of households (*inhabitants*) for C-PT refers to household wastewater connections only and excludes number of users from public toilet connection. Water supply for C-PT describes household water infrastructure only.

[b] Greywater is defined as "water generated from washing food, clothes and dishware, as well as from bathing, but not from toilets"; blackwater is defined as the mixture of urine, faeces and water used for flushing, along with cleansing material (Tilley et al., 2014).

3.3.2 Data collection

3.3.2.1 Biogas use, production and losses

Biogas use and production was monitored using diaphragm gas meters (G4 200, Elster Group, Germany) and pulse data loggers (UX90-001, Onset Computer Corporation; Bourne, MA). Gas meters, with attached loggers, were permanently installed at all fifteen ADs throughout the 6-month study period. Biogas use was recorded on an hourly basis over the entire 6-month study period *via* the connected data loggers. Biogas production was measured at approximately monthly intervals between March–August 2015 *via* controlled release of gas over a 48-hour period. Each device was positioned so as to avoid interfering with the household's cooking activities. Biogas losses were calculated as the difference between mean biogas production and measured biogas use *per* 24-hour period.

3.3.2.2 Wastewater treatment performance

Wastewater treatment performance was assessed by evaluating reduction in (1) chemical oxygen demand (COD) (mg/L) and (2) fecal indicator bacteria (FIB) (CFU/100mL). Data related to wastewater treatment were collected at only 13 of the 15 ADs because inflow pipes at 2 digesters were below water level. At each AD, COD measurements took place six to eight times (March–August 2015) and quantification on FIB took place three times (June–August 2015), each at approximately monthly intervals.

Chemical oxygen demand (COD)

COD reduction efficiency is calculated as (Equation (3.1)):

$$\mathrm{COD\ reduction\ (\%)} = \frac{\mathrm{COD_{influent}}\left(\frac{\mathrm{mg}}{\mathrm{L}}\right) - \mathrm{COD_{effluent}}\left(\frac{\mathrm{mg}}{\mathrm{L}}\right)}{\mathrm{COD_{influent}}\left(\frac{\mathrm{mg}}{\mathrm{L}}\right)}. \qquad (3.1)$$

Accurate measurement of influent COD is problematic as the concentration and flow of raw wastewater varies considerably throughout the day. Thus, influent COD was inferred using a mass balance calculation (Equation (3.2)):

$$\mathrm{COD_{influent}}\left(\frac{\mathrm{mg}}{\mathrm{L}}\right) = \mathrm{COD_{effluent}}\left(\frac{\mathrm{mg}}{\mathrm{L}}\right) + \frac{\mathrm{CH_{4\,(g)}} * \mathrm{f_{bg}}}{\mathrm{Q}} + \frac{\mathrm{CH_{4\,(l)}} * \mathrm{f_{bg}}}{\mathrm{Q}}. \qquad (3.2)$$

where $COD_{(influent)}$ = influent COD in mg/L, $COD_{(effluent)}$ = effluent COD in mg/L, Q = mean wastewater flow-rate in m^3/day, $CH_{4\ (g)}$ = CH_4 released in the gas phase in

m^3/day, $CH_{4\,(l)}$ = CH_4 that remains dissolved in the liquid effluent (m^3/day) and f_{bg} = a conversion factor for CH_4 to COD adjusted for site conditions.[20]

At each wastewater sampling event, $COD_{(effluent)}$ was measured with a one-liter grab sample of effluent wastewater collected at each AD outlet manhole. Samples were collected between 6 am to 6 pm and were immediately stored on ice. All samples were tested within 6 hours *via* photo spectrophotometry (Hach Lange Photometer (DR2800), heater (LT200) and test cells (LCI400, range: 0 – 1000 mg/L COD)). Dilution of samples using distilled water ranged from no dilution for low-strength wastewater to a maximum ratio of 1:5 for high-strength wastewater. The necessary dilution was estimated visually prior to testing. Duplicates of all samples were tested, with the mean used as the representative value of the sample.

The mean daily wastewater flow was measured during each sampling event using a 3L bucket and stopwatch. At each AD, three flow rate measurements of influent wastewater, spaced a minimum of two minutes apart, were taken every hour for a 48- to 96-hour period. The average of the three measurements was assumed to be the mean hourly flowrate, which was used to estimate the total volume of wastewater for the respective hour. The daily flow was then calculated as the sum of hourly volumes over a 24-hour measurement period.

Biogas CO_2 content was measured at all ADs during each wastewater sampling event using a CO_2 indicator (Brigon Messtechnik GmbH; Rodgau, Germany). Distribution of biogas amongst CH_4, CO_2 and N_2, as well as percentage of CH_4 dissolved in the liquid phase, was calculated as a function of influent COD concentration and

[20] The mass balance equation assumes that measured COD in the effluent is entirely biodegradable and that COD removed in settled sludge is negligible. While there is some accumulation of COD in the reactor, the long residence time (3-5 years) of the sludge allows for near-complete degradation of the biodegradable COD. Thus, the sludge accumulation rates in the ADs are principally due to accumulation of inert solids and non-biodegradable COD, which are neglected in this analysis. The mass balance approach therefore underestimates the total influent COD concentration and thus the calculated *per-capita* COD production and AD treatment efficiency.

wastewater temperature, following theoretical equilibrium methods described by Cakir and Stenstrom (2005) and McCarty et al. (2015). As influent COD concentration was inferred using a mass balance (Equation (3.2)), an iterative method, with linear interpolation between incremental influent COD concentrations, was used to simultaneously solve for influent COD and biogas partitions.

$CH_{4(g)}$ production (m^3/day) was computed using measured biogas production and theoretical equilibrium $CH_{4(g)}$ content. $CH_{4(l)}$ (m^3/day) production was then calculated using theoretical equilibrium $CH_{4(l)}$ content and $CH_{4(g)}$ production (Equation (3.3)):

$$\mathrm{CH}_{4(\mathrm{l})}\left(\frac{\mathrm{m}^3}{\mathrm{day}}\right) = \mathrm{CH}_{4(\mathrm{l})}(\%) * \frac{\mathrm{CH}_{4\,(\mathrm{g})}\left(\frac{\mathrm{m}^3}{\mathrm{day}}\right)}{1 - \mathrm{CH}_{4(\mathrm{l})}(\%)}. \tag{3.3}$$

Microbiological analysis

Concentrations of total coliforms (TC) and *Escherichia coli* (*E. coli*) were measured in both influent and effluent using the IDEXX Quanti-Tray/2000® system with Colilert® reagents (IDEXX Laboratories, Maine, USA). Composite influent wastewater samples were produced at each AD by collecting three 3L samples of influent wastewater each hour from all inflow pipes (according to the flowrate measurement described above). If the wastewater flow was too low to fill the 3L volume within a 5-minute interval, all wastewater within the interval was collected. Each hourly sample was homogenized for a minimum 30-second period using a battery-operated drill with a mixing paddle. A one-liter grab sample of this mixture was collected and immediately stored on ice. This process was repeated each hour for a 24-hour period at each AD. At the end of the 24-hour period, a composite sample was created by mixing a portion of each hourly sample in proportion to the relative flow measured for that hour.[21] Effluent

[21] Standard methods specify testing within 6 hours of sampling. However, a 24-hour sampling campaign was required to produce a representative daily influent flow. Others have shown bacterial populations are not significantly affected for periods up to 24 hours when stored at 5°C (Olanya et al., 2014). As samples were stored on ice for the duration of the 24-hour sampling period, we similarly expect that FIB concentrations were not significantly

wastewater was sampled at the end of the 24-hour influent sampling period: a one-liter grab sample was collected at each AD outlet and immediately stored on ice. All effluent wastewater samples were processed within 6 hours of collection.

Each influent sample was diluted with distilled water at 1:10^6 and 1:10^8; while each effluent sample was diluted at 1:10^4 and 1:10^6. Diluted samples were processed by adding a reagent pack (IDEXX) to the diluted 100mL sample. The sample/reagent mixture was then poured into sterile Quanti-Trays/2000 (IDEXX) and heat sealed (Quanti-Tray Sealer, IDEXX). The sealed trays were incubated at 35°C for 18 hours. After incubation, fluorescent wells were identified using a 6-watt, 365-nm UV light (IDEXX) and the number of TC and *E. Coli*, respectively, were calculated using an MPN table (IDEXX). The least diluted sample within the detection range was used for quantifying the MPN for influent and effluent samples.

3.3.2.3 Supporting field measurements

Additional data were collected to explain possible variation in AD performance. Air temperature and effluent wastewater temperature were measured hourly over the 6-month study period using Hobo 64K Pendant Temperature Data Loggers (Onset Computer Corporation). Two air temperature loggers fitted with solar radiation shields (Onset) were installed at each of the three study sites. One water temperature logger was installed at each AD outlet, positioned one meter below the water overflow level. Additionally, pH was measured at each AD outlet manhole at the time of wastewater sampling using a digital pH meter and field probe sensor (Ecosense pH100A, YSI, Yellow Springs, Ohio).

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affected. Sampling was also planned according to peak flow times so that high flow rates (and therefore a higher proportion of the composite sample) coincided with shorter storage times.

3.3.2.4 Household surveys and energy monitoring

Structured household surveys and energy use monitoring were carried out over a 3-week period in May 2015. A cross-sectional research design was used to evaluate differences in energy use and financial expenditure on cooking energy between households with and without biogas connections. In total, 120 households participated in the surveys (40 households *per* study site) including all 24 households with biogas connections and 96 without connections. Households without biogas were systematically sampled by identifying every other house on a community map (starting from those with biogas connections and then moving outward). If the occupant of a sampled household declined to take part in the survey, the adjacent household was requested to take part. In total, 8 households (6%) out of 128 households declined to participate.

The household survey was used to collect information on energy and sanitation perceptions and behaviors, as well as on socio-economic conditions. Surveys were performed by trained enumerators in teams of two people, with at least one person fluent in both the local language and English. Interviews were conducted primarily in the local language, with responses translated to English and recorded on paper during the interview. Responses were recorded in an electronic database on the same day as an interview was conducted. Mean interview time was 33 minutes (standard deviation (SD) = 7 minutes).

Daily measurements of cooking fuel use and financial expenditures on cooking fuel were collected over a five-day period at each household that participated in the survey. The primary cooking fuels used in the study communities are charcoal, electricity and biogas. Charcoal use was measured with a 5kg spring balance for smaller daily-use bags and with a 50kg spring balance for larger bulk bags. The difference in bag weight from one day to the next was assumed to equal charcoal use in the previous 24-hour period. Electricity use was measured with electricity appliance monitors (Energenie Power Meter, Sheffield, UK), which were installed continuously over the five-day period for all cooking devices (*e.g.*, stove and electric kettle) used by the household.

Biogas use was measured as described in the biogas production and consumption methods section. The mean price *per* kg of charcoal at each study site was determined *via* measurements of a variety of charcoal bags from community sellers and neighborhood markets. The price of electricity *per* kWh at each study site was obtained *via* receipts issued by the power company. At the time of the study, biogas connections, appliances and biogas supply were all provided to households free of charge.

Additionally, a brief survey (less than five minutes' duration) was conducted at all households in each community during May 2015 to determine the total number of people served by wastewater connections to ADs. The total population served was used to determine *per*-inhabitant wastewater, biogas and COD production rates.

3.3.3 GHG emissions and financial analysis

The incremental GHG emissions and financial costs and benefits of ADs were examined using a comparative analysis of wastewater treatment only *versus* wastewater treatment with energy recovery. For this analysis, septic tanks (STs) are assumed to be the conventional alternative to the ADs in operation at the three study sites. Similar to ADs, the primary treatment mechanism for STs is anaerobic digestion; in contrast, STs do not capture CH_4 produced *via* anaerobic digestion. The theoretical STs were modeled with an identical liquid reactor volume to each AD and therefore are assumed to achieve equal COD removal rates to each corresponding AD. By assuming similar COD removal and anaerobic conditions, equivalent rates of biogas production and partitioning are also assumed for both cases.[22] Conveyance and secondary treatment infrastructure are assumed to be similar and are therefore excluded from the analysis.

[22] The methane conversion factor (MCF) for a septic tank is listed as 0.5 based on a 50% reduction of BOD within a ST, implying anaerobic conditions similar to those within an AD (IPCC, 2006).

3.3.3.1 GHG emission analysis

GHG emissions associated with each AD and its corresponding theoretical ST are calculated considering wastewater treatment process emissions, recovery of CH_4 emissions, and offsets in conventional cooking energy use (Equation (3.4)). To allow for summation, all emissions are converted to carbon dioxide equivalents (CO_2e) using 100-yr global warming potentials (GWP) (IPCC, 2012).[23]

$$\mathrm{CO_2e_{(balance)}} = \mathrm{CO_2e_{(CH4,gas)}} + \mathrm{CO_2e_{(CH4,liquid)}} - \mathrm{CO_2e_{(CH4,recovered)}} - \mathrm{CO_2e_{(fuel\ offset)}}. \quad (3.4)$$

$CO_2e_{(CH4,\ gas)}$ and $CO_2e_{(CH4,\ liquid)}$ refers to the CO_2 equivalent of CH_4 production released in the gas and liquid phases, respectively. $CO_2e_{(CH4,\ liquid)}$ is assumed to be released upon effluent discharge from the AD and is therefore included in the CO_2e balance. These emissions are assumed to be equal for the ST and AD scenario.

$CO_2e_{(CH4,\ recovered)}$ refers to the CO_2 equivalent of CH_4 emissions recovered from treatment processes, which are combusted through energy reuse and converted to CO_2.[24] The resulting CO_2 emissions are considered to be biogenic and therefore do not contribute to the carbon balance (US-EPA, 2011). Thus, the CO_2e of recovered CH_4 emissions are subtracted in Equation (3.4).

$CO_2e_{(fuel\ offset)}$ refers to the offset in emissions resulting from differences in conventional fuel use found for households with biogas *versus* households without biogas supply. Emissions from charcoal and electricity use are considered in the analysis. For the AD scenarios, $CO_2e_{(fuel\ offset)}$ at each AD is calculated as the difference between mean *per-capita* charcoal and electricity use for the study-site population with *versus* without biogas connections, multiplied by the number of gas users *per* AD. The difference in fuel use is then converted to CO_2e according to

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[23] 100-yr GWP: CO_2 = 1; CO = 3.3; CH_4 = 28 (IPCC, 2013). The 100-yr GWP of CH_4 including climate-carbon feedbacks is reported as 34 by IPCC (2013); thus our estimates for CO_2e of CH_4 are likely conservative.

[24] Equation for CH_4 combustion: $CH_4 + 2O_2 \rightarrow CO_2 + 2\ H_2O$

emissions factors from literature sources (see Appendix A).[25] For the ST scenario, no difference in conventional fuel use is assumed.

3.3.3.2 Financial analysis

The financial viability of biogas recovery *via* communal ADs is assessed using a net present value (NPV) analysis of the AD *versus* ST scenarios. The analysis investigates whether the additional costs necessary to facilitate biogas storage, transport and utilization are financially justified. Thus, only differences in capital and recurrent costs between scenarios are considered. Costs and benefits that are equivalent across scenarios, such as the cost of desludging or potential revenue *via* a sewerage surcharge, are excluded from the analysis. The potential economic benefits of improved health resulting from provision of wastewater treatment and clean cooking energy are also excluded.

Capital costs (CapEx) for the ST and AD masonry structures were estimated at US $36 and US $42 *per* inhabitant, respectively (BORDA, 2016).[26] In the AD scenario, an additional CapEx of US $3 *per* inhabitant was estimated for gas piping and stoves (BORDA, 2016).[27] A 20-year lifespan was assumed for masonry structures in each

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[25] Accounting of emissions resulting from biomass fuel use is dependent on the consumption and regeneration cycle of the particular biomass under consideration (US-EPA, 2011). Typical practices for charcoal production in eastern and southern Africa reportedly result in the felling of entire trees and clear-cutting around kiln sites leading to transformation of vegetation structure and composition over large harvested areas (Chidumayo and Gumbo, 2012). Thus, in this study, we assume household charcoal use results in a long term decline in the total carbon embodied in standing biomass and a net release of carbon.

[26] CapEx includes construction materials, labor, technical design and project management but excludes costs for user interface, sewerage, secondary treatment infrastructure, land and community engagement for both AD and ST scenarios. CapEx for ADs were calculated using construction documents, while CapEx for theoretical ST alternatives were estimated assuming equal construction and material specifications apart from the rectangular shape of the tank and reinforced concrete top-slab. Each ST was sized according to the liquid volume of a corresponding AD, disregarding the gas volume as no gas storage is required in the ST case. All cost calculations were provided by BORDA (2016), a technical advisor for the sanitation project design and implementation.

[27] Each household gas connection was estimated at US $290 including materials, labor, technical design and project management. CapEx cost of US $3 refers to the *per*-inhabitant

scenario; a 6-yr lifespan was assumed for biogas piping and stoves (Gutterer et al., 2009). Annual operations and minor maintenance costs (OpEx) were estimated as 5% of CapEx in each scenario (BORDA, 2016).[28] The *per*-inhabitant CapEx and OpEx figures are within the range of values cited in the literature for primary treatment and septic tanks (Nelson and Murray, 2008; WASHCost, 2012).

Site-specific values for *per*-inhabitant biogas production, energy costs and carbon emissions are used to identify the conditions under which a AD option may be financially viable. A real discount rate of 5% is assumed for both AD and ST scenarios (World Bank, 2013).[29] Five financial scenarios with varying revenue generation *via* energy recovery and carbon financing are considered, as follows:

- Scenario 1: no payment for biogas connection or supply (current scenario),
- Scenario 2a: theoretical biogas tariff based on the mean daily use of CH_4 *per capita* (MJ/cap/day) and the price, energy content and types of conventional cooking fuels used in each study site,[30]
- Scenario 2b: theoretical biogas tariff according to the difference in mean financial expenditures on cooking fuels *per capita* for households with *versus* without biogas supply (measured *via* household energy surveys),
- Scenario 3a: carbon financing based on mean difference in CO_2e emissions *per capita* and cost of carbon at \$2.40/tCO_2e (Kossoy et al., 2015)[31] and

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cost spread over entire population served by sanitation system. The cost of gas piping and stoves were included in the AD case and excluded from the ST case.

[28] Annual OpEx costs include costs for primary treatment only and exclude costs incurred by user (e.g. cleaning products for household latrine) and costs for sewerage and secondary treatment. Direct support, indirect support or capital maintenance costs are not included for either scenario (Fonseca et al., 2011).

[29] Inflation rate is not considered in analysis as real discount rate is used.

[30] This calculation makes the assumption that prices of alternative energy fuels are a reasonable proxy for the price of biogas energy. Other households in the study area were observed to have made the transition from traditionally used charcoal to modern forms of energy such as electricity; the same transition can be reasonably expected for biogas.

31 Cost of carbon of \$2.40/tCO2e based on auctioned price of carbon under the Pilot Auction Facility for Methane and Climate Change Mitigation (PAF) mechanism in July 2015 (Kossoy et al., 2015).

- Scenario 3b: carbon financing based on mean difference in CO_2e emissions *per capita* and social cost of carbon (SCC) of \$36/tCO_2e (IWG, 2015).[32]

3.3.4 Ethics

The study was reviewed and approved by Stanford Human Subjects Research on 29 January 2015 (Protocol ID 33195; IRB Number 349). Informed oral consent was obtained from either the male or female head of each household participating in the surveys and energy use measurement.

3.4 Findings

3.4.1 Temperature and pH

Air temperature among the three study sites ranged from a mean high of 30.9°C (SD=3.4°C) in March to a mean low of 7.8°C (SD=3.1°C) in July. Less diurnal and monthly variability was observed for wastewater effluent temperatures, which ranged from a mean high of 26.0°C (SD=0.4°C) in March to a mean low of 21.9°C (SD=0.4°C) in July. The mean pH of AD effluent wastewater was 6.8 (SD=0.2), 6.9 (SD=0.1) and 7.0 (SD=0.2) at Sites A, B and C, respectively, indicating stable anaerobic conditions.

3.4.2 Anaerobic digester energy and wastewater treatment performance

Study site wastewater characteristics and AD treatment and energy performance at each site are reported in Table 3.2.[33] Digester C-PT is tabulated individually, as its

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[32] The SCC is an estimate of the economic damages associated with an incremental increase of one metric ton of CO_2 in a given year, or as the value of damages avoided for the reduction of one metric ton of CO_2 (IWG, 2015).

[33] Production rates are typically described in the literature in terms of '*per capita*' to describe the physical quantity of waste or wastewater produced *per* person. We depart from conventional terminology and instead use '*per* inhabitant' to describe mean rates discharged into (or generated from) ADs for the population with household wastewater connections. *Per*-inhabitant rates may be greater than *per-capita* rates if, for example, additional organic waste is discharged into the ADs, or may be lower if not all waste produced by the population served is discharged into the ADs.

connection to a public toilet results in its inflow characteristics and performance differing considerably from the other ADs at Site C. Notably, nearly all households at Sites A and B, while less than half of households at Site C, report use of sewer connected toilets as their primary facility. Mean wastewater, biogas and COD generated *per* inhabitant were found to be significantly higher at Site A as compared to Sites B and C (all $p<0.01$). Although mean wastewater generated *per* inhabitant was significantly higher at Site C compared to Site B, mean biogas and COD production *per* inhabitant were both significantly higher at Site B. Biogas and COD production *per* inhabitant appear to be remarkably high for digester C-PT; however, the number of inhabitants reported captures only the population with household toilet connections and excludes public toilet users. COD and biogas production rates measured at Site A are similar to rates reported in the literature for developing country contexts, while rates observed at Site B and C are generally lower than literature values (Mang and Li, 2010; Lohri et al., 2010; Reynaud and Buckley, 2015).

Regular use of biogas was reported by all gas-connected households, with similar rates of consumption *per* user across study sites. *Per*-inhabitant production of biogas ranged from 4-13% of *per-capita* consumption by gas-users. Loss of biogas resulting from unused biogas production was considerable: mean biogas losses were 19% (SD=15%) of total production at Site A, 20% (SD=26%) at Site B and 11% (SD=25%) at Site C.[34] Although mean biogas consumption *per* user was considerably higher for digester C-PT as compared to other sites, biogas losses resulting from unused biogas production accounted for 55% (SD=4%) of total production.

Mean influent COD concentration varied considerably across sites, with higher concentrations at Site B and digester C-PT in comparison to Sites A and C. COD reduction was found to range from 61%-83%, with higher reduction efficiencies at

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[34] When the gas volume in fixed dome digesters reaches the maximum gas storage capacity, any further biogas production results in loss of biogas released directly into the environment as CH_4 and CO_2.

higher COD influents at Site B and digester C-PT. Reduction of TC and *E. Coli* ranged from 0.9 to 1.3-log and 0.5 to 0.9-log, respectively.

Inferred CH_4 content was found to vary inversely to influent COD concentration, ranging from 72-76% at higher COD concentrations found at BGD C-PT and Site B, respectively, to 82% at the lower COD concentrations found at Sites A and C. Inferred CO_2 content follows the general trend of measured values at varying COD concentrations but is calculated at 3-8% higher than measured values.

Table 3.2: Wastewater characteristics and anaerobic digester (AD) performance *per* study site

Parameter	Site A [a, b]			Site B			Site C [a, b, c]			C-PT [d]		
	n	**Mean**	**(SD)**	***n***	**Mean**	**(SD)**	***n***	**Mean**	**(SD)**	***n***	**Mean**	**(SD)**
Percentage of households reporting use of sewer connected toilet as primary facility (%) [e]	*83*	100%	-	*91*	99%	-	*248*	48%	-	*15*	40%	-
Reactor volume *per* inhabitant (m^3/inh.) [f]	-	0.26	-	-	0.15	-	-	0.22	-	-	0.17	-
Wastewater generation *per* inhabitant (L/inh./day) [g, h]	*17*	74	(10)	*14*	13	(4)	*16*	17	(4)	*15*	21	(4)
Hydraulic retention time (days) [g]	*17*	4.5	(2.8)	*14*	14.3	(6.9)	*16*	9.1	(6.6)	*15*	8.3	(1.8)
Organic loading rate (OLR) (kgCOD/m^3 reactor/day) [g]	*17*	0.26	(0.15)	*14*	0.20	(0.04)	*16*	0.12	(0.05)	*15*	0.58	(0.11)
Biogas production *per* inhabitant (L/inh./day) [h, i]	*12*	20	(1)	*8*	12	(2)	*8*	3	(0.4)	*12*	50	(12)
Biogas consumption *per* gas user (L/gas user/day) [g, i]	*182*	155	(10)	*182*	137	(27)	*176*	132	(20)	*176*	288	(52)
Biogas losses *per* inhabitant (L/inh./day) [g, i]	*182*	4	(1)	*182*	3	(2)	*176*	0.5	(0.3)	*176*	28	(8)
CO_2 content of biogas (measured) (%) [i, l]	*24*	10%	(2%)	*12*	15%	(4%)	*49*	10%	(4%)	*7*	21%	(1%)
CO_2 content of biogas (inferred) (%) [i, k]	*24*	13%	(3%)	*12*	23%	(4%)	*49*	13%	(4%)	*7*	28%	(1%)
N_2 content biogas (%)[i, k]	*24*	4%	(1%)	*12*	1%	(1%)	*49*	5%	(3%)	*7*	1%	(0.1%)
CH_4 content of biogas (%) [i, k]	*24*	82%	(1%)	*12*	76%	(3%)	*49*	82%	(1%)	*7*	72%	(1%)
Percent of CH_4 production dissolved in effluent (%) [i, k]	*24*	12%	(3%)	*12*	4%	(2%)	*49*	13%	(6%)	*7*	2%	(0.4%)
COD, influent (mg/L) [i, k]	*24*	851	(257)	*12*	2708	(1215)	*48*	830	(346)	*7*	4766	(1073)
COD, effluent (mg/L) [i, l]	*24*	323	(67)	*12*	812	(207)	*48*	327	(84)	*7*	811	(77)
COD production *per* inhabitant (g/inh./day) [i, h, k]	*24*	62	(12)	*12*	33	(11)	*48*	13	(5)	*7*	97	(18)
Total coliforms, influent (Log CFU/mL) [i, l]	*9*	8.0	(7.9)	*5*	8.1	(7.8)	*20*	7.9	(7.8)	*3*	8.6	(8.0)
Total coliforms, effluent (Log CFU/mL) [i, l]	*9*	7.1	(6.5)	*4*	7.1	(6.9)	*20*	7.0	(6.7)	*3*	7.3	(6.8)
E. Coli, influent (Log CFU/mL) [i, l]	*9*	7.4	(7.1)	*4*	7.7	(7.5)	*20*	7.6	(7.7)	*3*	8.2	(7.9)
E. Coli, effluent (Log CFU/mL) [i, l]	*9*	6.8	(6.5)	*4*	6.8	(6.7)	*20*	6.8	(6.7)	*3*	7.4	(7.4)

[a] ADs A-2 and C-4 excluded from wastewater analysis due to inflow pipe below water level preventing measurement
[b] *Per*-inhabitant figures have been adjusted for ADs connected in series for Sites A and C
[c] Site C analysis excludes digester C-PT
[d] Number of inhabitants for digester C-PT refers to people with household toilet connections only (excludes public toilet users)
[e] n refers to number of survey respondents in community-wide survey

[f] Reactor volume, HRT and OLR based on total liquid plus sludge volume of AD (decrease in reactor volume due to accumulated sludge is not considered)
[g] n refers to number of days of measurements
[h] Wastewater generation, biogas production, biogas losses and COD production based on number of inhabitants
[i] n refers to number of direct or inferred measurements
[j] Biogas use based on number of gas users (people obtaining biogas supply)
[k] Inferred values based on mass balance (Equation (3.2)). A methane conversion factor (f_{bg}) is calculated for each site using the Ideal Gas Law and f_{bg}=1/385 gCOD/ml-CH_4 at 20°C (Soto et al., 1993). Based on these assumptions, f_{bg}=1/456 at mean temperature of 22°C and 1350m elevation for Site A, 1/453 at 24°C and 1250m elevation for Site B, and 1/437 at 25°C and 920m elevation for Site C.
[l] Measured value

3.4.3 Household surveys and energy use

In total, 93 (78%) surveys were conducted with the female head of household, 22 (18%) were conducted with the male head and 5 (4%) were conducted with both the female and male head.

3.4.3.1 Household characteristics

At all sites, households with and without biogas connections are similar with respect to most measured socio-economic and demographic characteristics (Table 3.3). A notable exception is the mean number of people *per* household, found to be significantly smaller for those with biogas connections at Sites A ($p=0.01$) and C ($p<0.01$), and for the full sample ($p<0.01$). As households were selected for biogas connections primarily based on their proximity to AD infrastructure, the reason for this difference in household size is unclear. We analyze energy use and financial expenses on a *per-capita* basis to account for the systematically different household sizes across groups.

In contrast, significant differences in socio-economic characteristics were found across study sites. Site A inhabitants have significantly higher mean education levels, greater mean number of rooms *per* household, ownership of household assets and are significantly more likely to be employed in the formal *versus* informal sector as compared to both Site B and C inhabitants (all $p<0.01$). Compared with inhabitants in Site B, Site C inhabitants have a significantly greater mean number of rooms *per* household ($p=0.02$), are more likely to have electricity service ($p<0.01$), own a television ($p=0.08$) and are more likely to be employed in the formal *versus* informal sector ($p=0.04$).

Table 3.3: Socio-economic and demographic characteristics of households with *versus* without biogas (BG) supply

Parameter		**SITE A**		**SITE B**		**SITE C**		**OVERALL**	
		W/ BG	NO BG	W/ BG	NO BG	W/ BG	NO BG	W/ BG	NO BG
		(n=10)	*(n=30)*	*(n=5)*	*(n=35)*	*(n=9)*	*(n=31)*	*(n=24)*	*(n=96)*
Number of people *per* household [a]	Mean	4.1	5.9**	5.6	5.1	3.9	6.3***	4.3	5.7***
	(SD)	(1.7)	(2.2)	(2.3)	(2.3)	(1.5)	(1.9)	(1.8)	(2.2)
Years of education: male head of household	Mean	14.5	12.5*	N/A [c]	8.5	7.6	8.7	11.3	9.9
	(SD)	(1.6)	(3.4)	N/A [c]	(3.4)	(5.5)	(4.0)	(5.2)	(4.0)
Years of education: female head of household	Mean	11.4	11.2	6.0	5.0	6.0	7.0	8.0	7.6
	(SD)	(2.6)	(2.0)	(4.7)	(3.8)	(4.2)	(3.2)	(4.5)	(4.0)
Number of rooms *per* household [b]	Mean	5.6	5.1	2.2	2.7	3.4	3.0	4.1	3.5
	(SD)	(1.5)	(1.2)	(0.8)	(1.1)	(1.3)	(0.5)	(1.9)	(1.4)
% of households with electricity [d]	Mean	100%	100%	40%	43%	100%	100%	88%	79%
% of households reporting ownership of…									
…cellphone(s)	Mean	100%	100%	80%	71%	78%	87%	83%	86%
…television(s)	Mean	80%	77%	0%	6%	11%	19%	38%	32%
…motor vehicle(s)	Mean	40%	43%	0%	0%	0%	0%	17%	14%
Head of household employed in formal sector (%) [e]	Mean	80%	63%	0%	9%	22%	26%	42%	31%

T-test: 2-tailed test of means for households with *versus* without biogas: *0.05<p≤0.10; **0.01<p≤0.05; ***p≤0.01
n = number of household surveys *per* group
[a] Population figures exclude children less than 1-year-old for energy use and energy cost analysis
[b] Includes kitchen, dining, kitchen, bedroom(s) only
[c] No male heads of household
[d] All electricity supplied by grid connections; no other source of electricity reported
[e] Remaining % of head of households report informal sector employment (e.g. construction piecework, market trader, small-scale farmer, etc.)

3.4.3.2 Energy use

Sample households reported using charcoal, electricity and biogas as cooking fuels, with more than half of households reporting multiple energy use. On average across study sites, households with biogas connections used significantly less charcoal ($p<0.01$) and electricity ($p=0.01$) compared to those without biogas (Table 3.4).

We also compared cooking energy use for the two groups after converting all fuels to mega joules (MJ) using the energy content of each fuel type used (see Appendix A). Considering all fuel types combined, households with biogas used 29% less total energy for cooking *per capita* as compared to those without biogas ($p<0.01$). This finding is driven by the significantly large difference in cooking energy used by households with *versus* without biogas in Site B ($p<0.01$). Across sites, biogas use accounted for 41% of total cooking energy use among households with biogas connections (Figure 3.2). Charcoal was the primary conventional source of cooking energy for study households without biogas connections, accounting for 75%, 99%, 96% and 91% of total cooking energy at Sites A, B, C and overall, respectively. Electricity use was more common at Site A, accounting for 25% of household cooking energy use.

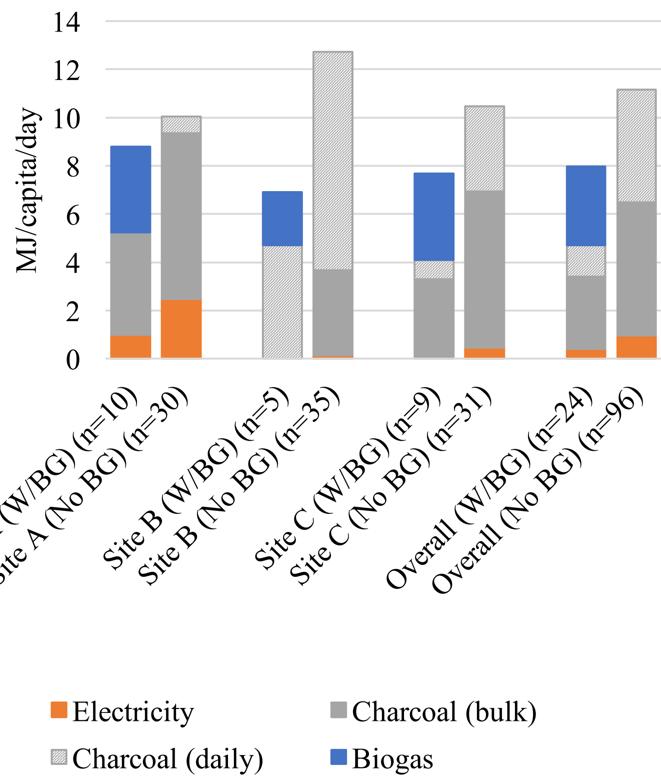
3.4.3.3 Energy expenditures

Charcoal prices across study sites ranged from US \$0.22-\$0.26 *per* kg for daily-use bags to US \$0.14-\$0.18 *per* kg for bulk bags (>20kg) (see Appendix A). Electricity prices were higher at Site A (\$0.04 *per* kWh) as compared to Sites B and C (\$0.02 *per* kWh). At all study sites, households with biogas reported significantly less financial expenditures on cooking energy *per capita* as compared to those without biogas (Table 3.4). Importantly, households reported no expenditure on biogas, as it was supplied free of charge at the time of the study. The theoretical expenditure on biogas used at study sites A, B, C and overall, based on prices and energy contents of conventionally used fuels, were estimated as \$0.03, \$0.02, \$0.02 and \$0.02 *per capita per* day, respectively. Even when incorporating this theoretical expenditure for biogas use, households with biogas still spend significantly less on cooking energy compared to households without biogas at Site B ($p<0.01$), C ($p=0.04$) and overall ($p<0.01$) (Figure 3.3).

Table 3.4: *Per-capita* energy use and financial expenditures for households with *versus* without biogas (BG) supply

Parameter		SITE A		SITE B		SITE C		OVERALL	
		W/ BG	NO BG	W/ BG	NO BG	W/ BG	NO BG	W/ BG	NO BG
		(n=10)	*(n=30)*	*(n=5)*	*(n=35)*	*(n=9)*	*(n=31)*	*(n=24)*	*(n=96)*
Cooking fuel use									
Charcoal (kg/*capita*/day)	Mean	0.16	0.29*	0.15	0.40***	0.14	0.35**	0.15	0.35***
	(SD)	(0.16)	(0.30)	(0.05)	(0.25)	(0.21)	(0.17)	(0.16)	(0.25)
Electricity (kWh/*capita*/day)	Mean	0.27	0.69***	0.00	0.04	0.00	0.12***	0.11	0.27**
	(SD)	(0.25)	(0.56)	(0.00)	(0.15)	(0.00)	(0.20)	(0.21)	(0.45)
Methane (L/*capita*/day)	Mean	127	-	90	-	131	-	116	-
	(SD)	(77)	-	(20)	-	(82)	-	(71)	-
Combined cooking energy use [a]									
Energy use (MJ/*capita*/day)	Mean	8.8	10.0	6.9	12.7***	7.4	10.5	7.9	11.2***
	(SD)	(4.1)	(7.0)	(1.8)	(7.7)	(5.6)	(4.6)	(4.4)	(6.7)
Financial expenditures on cooking fuels [b]									
Costs (USD/*capita*/day)[c]	Mean	$0.04	$0.08***	$0.04	$0.10***	$0.02	$0.06***	$0.03	$0.08***
	(SD)	($0.03)	($0.05)	($0.01)	($0.07)	($0.03)	($0.03)	($0.03)	($0.05)

T-test: 2-tailed test of means for households with *versus* without biogas: *0.05<p≤0.10; **0.01<p≤0.05; ***p≤0.01
n = number of household surveys *per* group
[a] Combined cooking energy use calculated by converting charcoal, electricity and methane cooking fuel use to MJ
[b] Theoretical cost of biogas not included
[c] Exchange rate: $1 USD = 7.25 ZMW (May 2015, oanda.com)



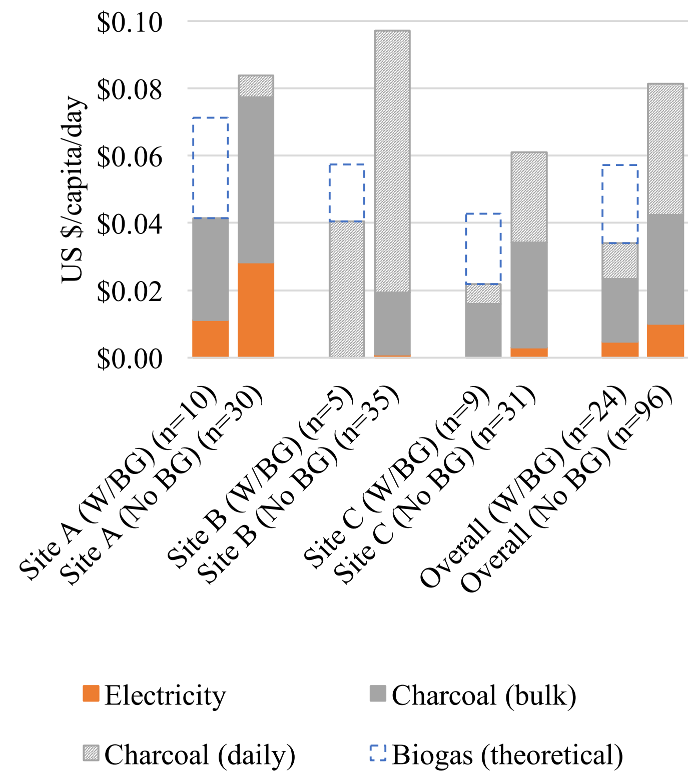


Figure 3.2: Mean daily *per-capita* energy use (MJ/*capita*/day), by study site and fuel type

Figure 3.3: Mean daily *per-capita* energy costs (US $/*capita*/day), by study site and fuel type

3.4.4 GHG emissions and financial analysis

3.4.4.1 GHG emissions

Across all study sites, mean *per*-inhabitant CO_2e emissions (Equation (3.4)), were found to be lower for the AD scenarios as compared to the corresponding theoretical STs (Table 3.5). A negative balance in emissions, as found for AD scenarios at Sites A–C, indicates that the sum of recovered CH_4 emissions and averted conventional fuel emissions is greater than CH_4 emissions released in the gas and liquid phases. Although emissions were positive for both the AD and ST scenarios for digester C-PT due to the substantial underutilization of biogas, the AD scenario represents an 80% reduction in CO_2e emissions *versus* the ST scenario. On average across sites, the mean difference in CO_2e emissions for the AD *versus* ST scenarios was 251gCO_2e/inh./day (SD = 131gCO_2e/inh./day). Analyses for GHG emissions and NPV were repeated using medians to address skewness in some indicators, with no substantive impact on results.

Table 3.5: Greenhouse gas (GHG) emissions for septic tank (ST) *versus* anaerobic digester (AD) scenarios

Parameter	**SITE A** *n = 32*		**SITE B** *n = 12*		**SITE C** *n = 56*		**C-PT** *n = 7*	
	Mean	(SD)	Mean	(SD)	Mean	(SD)	Mean	(SD)
CO_2e emissions (gCO_2e/inh./day) associated with...								
…CH_4, gas (production in gas phase)[a]	248	(44)	158	(68)	52	(26)	601	(142)
…CH_4, liquid (production in liquid phase)[a]	33	(11)	5	(1)	7	(3)	11	(2)
…CH_4, recovered [a]	199	(40)	131	(65)	43	(21)	268	(47)
…averted conventional fuel use	187	(145)	115	(5)	41	(46)	199	-
BALANCE: ST scenario (gCO_2e/inh./day)	250	281	(51)	163	(68)	59	(27)	612
BALANCE: AD scenario (gCO_2e/inh./day)	-110	-105	(152)	-82	(50)	-24	(48)	145

[a] CH_4 density (adjusted according to Ideal Gas Law, based on site elevation and mean temperature during study period): Site A: 0.58kg/m^3; Site B: 0.57kg/m^3; Site C: 0.60kg/m^3

3.4.5 Financial analysis

The financial viability of biogas recovery *via* ADs varies considerably across sites under a range of tariff and carbon pricing scenarios and considering differences in biogas production and financial expenditures on energy (Table 3.6). In the current scenario (1), in which households are not charged for biogas supply, the *per*-inhabitant NPV was negative US $16 across all study sites. In other words, this value represents the additional capital and operational cost of facilitating biogas energy recovery from wastewater treatment in this context. The NPV was also negative across all sites at the lower biogas tariff, while at the higher biogas tariff considered, positive NPVs were found at Sites A and B where higher *per*-inhabitant biogas production rates were measured. Two scenarios of carbon financing were also considered: across all sites negative NPVs are found at the lower carbon price (3a), while positive NPVs are found at the higher price (3b). The general trend of these findings are consistent for discount rates up to 9%.

Table 3.6: Net present value of additional investment and recurrent costs required for anaerobic digesters (relative to septic tanks)

Scenario		**NPV (US$ *per* inhabitant)** (5% real discount rate)			
		Site A	**Site B**	**Site C**	**Overall**
1	No biogas tariff (current condition)	($16)	($16)	($16)	($16)
2a	Biogas tariff based on energy content of biogas use	($2)	($11)	($14)	($9)
2b	Biogas tariff based on measured financial expenditures	$4	$1	($12)	($2)
3a	Carbon financing (at US $2.40/t$CO_2$e)	($12)	($13)	($15)	($13)
3b	Carbon financing (at US $36/t$CO_2$e)	$47	$24	$4	$25

3.5 Discussion and conclusions

The number of people living in SSA is projected to double by mid-century, with 70% of growth taking place in peri-urban areas with limited public infrastructure (UN-DESA, 2015). In such settings, and in the context of the Sustainable Development Goals (SDGs) and 2015 Paris Agreement on Climate Change, communal anaerobic digesters (ADs) are worthy of consideration as an alternative to conventionally used septic tanks (STs) for primary treatment of wastewater. Within this study, we found treatment efficiencies of ADs, ranging from 61-83% COD removal, are comparable to or greater than efficiencies of STs, reported in the literature to range from 25-50% (UNEP, 1997; Sasse, 1998; Foxon, 2009).

Understanding the costs and benefits of biogas recovery from domestic wastewater is an important contribution towards the realization of international and country specific development goals. Investment in ADs may simultaneously contribute to multiple SDGs including goal #6, which aims to achieve universal access to safely managed sanitation (target 6.2) and halve the proportion of untreated wastewater globally (target 6.3); goal #7, which aims to achieve universal access to modern, clean fuels (target 7.1) and substantially increase the share of renewable energy in the global energy mix (target 7.2); and goal #13, which includes integration of climate change measures into national policies, strategies and planning including fostering low emissions development (target 13.2) (UN, 2015). Moreover, Zambia's intended nationally determined contribution (INDC) to the 2015 Paris Agreement aims to reduce emissions by 25% by 2030 and targets domestic wastewater for climate adaptation efforts, including identifying biogas plants as a mitigation solution (INDC Zambia, 2015).

Our findings suggest that ADs have the potential to offer greater GHG emission reductions and financial benefits relative to STs. In particular, ADs appear promising in settings with (1) adequate water access, (2) conventional use of low efficiency fuels and (3) adequate demand for biogas in proximity to supply. Additionally, a financial mechanism to generate revenue from biogas recovery is necessary for favorable investment in ADs relative to STs.

Convenient and reliable water access for households is critical to realizing the potential GHG emission, energy recovery and financial benefits of sewer-connected ADs. In this study, water access was found to influence *per*-inhabitant wastewater generation and usage rates of sewer-connected toilets. In turn, these factors influence the effectiveness of waste conveyance in sewer networks, the rate of organic matter discharged into ADs and ultimately the volume of biogas captured.

For example, with in-house piped water supply, Site A residents reported exclusive use of sewer-connected toilets resulting in significantly higher rates of *per*-inhabitant wastewater, COD and biogas production compared to other sites. In contrast, although Site B residents also reported exclusive use of sewer-connected toilets, limited and intermittent water supply resulted in low wastewater generation, poor conveyance and partial degradation of waste in the sewer network and loss of biogas prior to discharge in ADs. At Site C, over half of survey respondents reported primary use of pit latrines, rather than sewer-connected toilets, citing unreliable and inconvenient water supply as the primary reasons for this continued practice.

Our findings also suggest that benefits of ADs will be greatest in settings where households primarily use low efficiency fuels, such as charcoal. Recognizing the limitations of our cross-sectional study design, we found that, on average, households with biogas used 29% less total energy for cooking *per capita* compared to households without biogas. In contrast, when relative fuel efficiencies of biogas, charcoal and electricity are considered, no significant difference in mean cooking energy use was found between households with and without biogas (Kaoma and Kasali, 1994; Anozie et al., 2007).

Lower energy use by households with biogas also translated to greater financial savings than expected. At the time of the study, biogas was provided free of charge; unsurprisingly households with biogas connections spent less money on cooking fuel. However, the magnitude of cost savings is greater than expected assuming households would save amounts proportional to the energy content of biogas used. In fact, across study sites, households were found to save approximately twice as much *per capita* compared to theoretical savings.

Baseline fuel type also drives the magnitude of potential GHG emission reductions realized *via* use of ADs. In this study, the primary use of charcoal resulted in averted fuel emissions comprising half of CO_2e emission reductions for AD *versus* ST scenarios.

Averted emissions would be approximately 70% less if the conventional fuel replaced were liquid propane gas (LPG) and would be negligible for renewably harvested firewood or electricity generated *via* solar, wind or hydropower from non-tropical reservoirs. Conversely, averted emissions would be 20–50% higher for electricity generated *via* coal or hydropower from tropical reservoirs (Demarty & Bastien, 2011; Raadal et al., 2011).[35]

Realizing the environmental and financial benefits of biogas capture also depends on the willingness and ability of households to use it. On average across ADs in this study, 82% (SD=15%) of CH_4 produced was captured and used by gas connected households; this comprised the remaining half of CO_2e emission reductions for the AD *versus* ST scenarios. As the unused balance of biogas production is released directly as CH_4, further GHG reductions could be achieved by increasing the proportion of used CH_4. However, several challenges exist with spatial optimization of biogas supply, which is dependent on wastewater collection points, and demand for biogas including (1) higher production of gas in locations with insufficient demand and, additionally, physical and financial constraints of transporting gas; (2) biogas produced solely from domestic wastewater is insufficient to meet the cooking energy needs of a typical household, thus only a limited number of households within a community can be supplied with biogas; (3) social or cultural opposition to use of products derived from human waste; and (4) variability in biogas production – for example, due to variations in temperature and influent wastewater – as well as variability in demand – for example, due to transient community populations and variations in daily cooking energy use. Although the percentage of biogas use found in this study generally exceeds literature values, the extent of biogas losses nevertheless highlight the challenge of spatial planning to ensure proximity of biogas supply and demand for decentralized ADs as also noted by others (Tilmans et al., 2014).

[35] Emissions from firewood harvested renewably are biogenic and therefore have a net zero carbon impact (World Bank, 2001). Conversely, charcoal production typically results in felling entire trees and is considered to have a net positive impact on carbon emissions (Chidumayo and Gumbo, 2012).

Considering variations in water, baseline fuel and percentage of biogas use within this study, the mean balance of CO_2e emissions for ADs were negative across all sites indicating that, within this context, recovery of energy from wastewater has the potential to yield a net positive GHG impact. Our findings indicate that the use of ADs relative to STs may reduce annual *per-capita* emissions by 45–141 $kgCO_2e$, representing a reduction of 4–13% of the total estimated emissions produced *per capita* in Zambia.[36]

Finally, revenue generation from biogas supply, for example *via* a biogas tariff or carbon financing, is necessary for financial favorability of ADs relative to STs. Within this study, the minimum biogas tariff required for financial favorability of ADs *versus* STs ranges from US $0.22 to $1.86 *per* m3 biogas used, while the minimum price of carbon required ranges from US $9 to $28 *per* tCO_2e, for the more favorable conditions found at Site A to the less favorable conditions found in Site C, respectively.

Investment in alternative energy such as biogas recovery from ADs will become increasingly favorable with growing demand for energy. The minimum biogas tariff required for financial favorability of ADs within this study was cost competitive under more favorable site conditions – 27% less than the price of charcoal and similar to electricity *per* MJ – and substantially higher than conventional fuel prices under less favorable site conditions – 10 to 11 times higher than the price of charcoal and electricity *per* MJ, respectively. At the time of this study, charcoal was relatively inexpensive due to Zambia's low population density and limited enforcement of charcoal production regulations (World Bank, 2015; USAID, 2010). Government subsidized electricity tariffs were also among the lowest in the world (AfDB, 2013). However, with a four-fold increase in Zambia's population expected by mid-century and one of the highest rates of deforestation in the world, increasing demand for wood-fuel, coupled with diminishing supply, will drive charcoal prices up (UN-REDD, 2012; UN-DESA, 2015). Additionally, progressive increases to electricity tariffs are underway to reach cost-reflective levels beginning with a 75% increase proposed in 2017 (Bloomberg, 2016; Zambian Politics,

[36] Combined annual CO_2 and CH_4 emissions *per capita* in Zambia are estimated at 1090 $kgCO_2e$ excluding land-use change and forestry (WRI, 2013).

2017). Moreover, increasing climate variability will affect the reliability and output of electricity supply in Zambia, which relies almost entirely on hydropower, driving up demand and prices of alternative fuels (ERB, 2015). Indeed, since the time of data collection for this study, several years of regional drought have led to severe electricity deficits resulting in nationwide rolling blackouts and marked increases in charcoal prices (GPJ, 2015; NYT, 2016).

Current prices in global carbon markets are not conducive to investments in biogas recovery from wastewater. At present, the lower-bound carbon price used in this study is indicative of available financing levels for development projects. However, some studies have suggested carbon prices higher than the upper-bound used in this study are required to reflect the true external costs of carbon emissions (Howard, 2014; Moore and Diaz, 2015). Additionally, the United Nations Global Compact (UNGC) has called for a minimum carbon price of US \$100/tCO_2e by 2020 (World Bank and ECOFYS, 2016). At this price, considering mean CO_2e emission reductions across sites, the additional investment required to facilitate biogas recovery from ADs yields a positive *per-capita* NPV of US\$ 91 and a simple pay-back period of less than one year. Moreover, considering total capital and operating costs, a positive *per-capita* NPV of US\$ 32 is found for ADs within this study. In other words, at this level of carbon pricing, biogas recovery could offset the total capital and recurrent costs of primary wastewater treatment.

3.5.1 Implications

The proposed sanitation target of the SDGs is to achieve universal access to sanitation by 2030 (UN, 2015). Given that by 2015, 44% of urban residents in Zambia lacked access to improved sanitation and the urban population is expected to nearly double to 12 million people by 2030, an additional 8.5 million people in urban Zambia will require sanitation infrastructure to meet the SDG target (UN, 2015; UN-DESA, 2015).

Our findings suggest that with concurrent implementation of water supply infrastructure to realize the full benefits of sewer-connected ADs, use of such infrastructure for primary wastewater treatment for this population could potentially reduce annual charcoal use by

41,000 tonnes, annual electricity use by 31 gigawatt-hours and offset 1.2 million tCO_2e emissions *per* year, relative to the use of STs. Considering the range of biogas tariffs and carbon prices assumed in the present study, the potential annual revenue that could be accessed through biogas recovery ranges from US $9.5 to $13.7 million *via* a biogas tariff and US $2.9 to $42.9 million *via* carbon financing. At the proposed UNGC price of $100/tCO_2e$, annual carbon financing could be up to US$ 119 million.

Mechanisms to ensure carbon pricing stability at these levels are necessary to attract investment in emissions reduction projects such as ADs. Zambia's INDC, budgeted at US$ 50 billion, is contingent upon securing funding for proposed climate adaptation and mitigation efforts (Zambia INDC, 2016). With smart financing models and robust carbon markets, under the conditions identified in this study, investment in ADs with biogas recovery could effectively contribute towards this climate goal while simultaneously contributing towards the SDGs of universal access to sanitation and clean energy.

3.5.2 Limitations

Our findings are limited to one study during one year in Zambia. Although this study can help to identify the conditions under which greater benefits are likely to be realized from ADs, the extent of costs and benefits of AD investment in other locations remains unclear. This study is also limited by a small sample size and a cross sectional research design. Although efforts were made to systematically sample households, differences between groups likely exist. Furthermore, as biogas is currently supplied at no cost, our financial analysis is based on theoretical tariffs rather than actual willingness to pay. Energy use and GHG emission reductions may differ from the figures reported if a tariff were to be introduced. Fuel use measurements are also limited to a specific time of year, whereas fuel use patterns and expenditures are likely to vary by season. Finally, our findings are contingent upon functioning AD and biogas piping infrastructure, whereas failure rates of ADs in the range of 50 - 75% have been reported in sub-Saharan Africa (Bond and Templeton, 2011). Identifying incentives to ensure greater numbers of ADs succeed will be critical to realizing potential environmental and financial benefits of ADs.

3.6 Acknowledgements

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4 Assessing the lifecycle resource use, greenhouse gas emissions and financial costs and benefits of sewered and non-sewered sanitation approaches: Evidence from Zambia and India

This chapter is being prepared as a manuscript. Susmita Sinha will be co-author for contributions in data acquisition, interpretation and manuscript revisions. Chris Buckley, Jennifer Davis and Michael Lepech will be co-authors for contributions to study design, modeling, data interpretation and manuscript revisions.

4.1 Abstract

Cities in sub-Saharan Africa (SSA) and South Asia (SA) have some of the highest rates of population growth and lowest rates of urban sanitation coverage in the world. Major infrastructure investments are required for rapidly growing unserved populations in these regions to achieve the Sustainable Development Goal (SDG) target of universal access to 'safely managed sanitation services' (WHO/UNICEF, 2015a). In recent years, decentralized sewered and non-sewered approaches have gained traction as alternatives to expensive and resource intensive conventional wastewater treatment. However, little evidence is available regarding the long-term costs and benefits of these new approaches.

In this study, a lifecycle framework is developed to quantify resource use, global warming potential (GWP), and financial costs and benefits of decentralized sewered and non-sewered sanitation approaches. Secondly, use of this framework is demonstrated using primary and secondary data collected from four operational sanitation systems located in India and Zambia. Findings from this comparative analysis suggest (1) a system-wide, rather than process, approach is essential to understand and compare approaches; (2) significantly higher water use for decentralized sewered systems *versus* non-sewered systems; (3) the percent of biogas recovery and baseline fuel use strongly influence lifecycle energy use and GWP for sewered systems, whereas biogas recovery had little influence on these outcomes for non-sewered systems; (4) capital infrastructure and water supply are important drivers of lifecycle energy use (37 – 77% and 31 – 56% of lifecycle energy use, respectively); (5) direct methane emissions drive the magnitude of GWP impacts (67 – 87% of lifecycle GWP); (6) conveyance of waste for non-sewered

systems accounts for moderate lifecycle energy use (18 – 24%) and minimal GWP impacts (1 – 6%); (7) significantly higher user and agency costs are likely for sewered systems *versus* non-sewered systems; however, non-sewered system users may incur higher upfront capital costs and a greater proportion of lifecycle costs compared to sewered system users. The framework presented in this study can be applied to other alternative sewered and non-sewered sanitation approaches to evaluate resource use, environmental impacts and financial costs and benefits, as well as the extent to which resource recovery may impact costs and benefits.

4.2 Introduction

By mid-century, the global urban population is projected to increase by 2.4 billion people with 60% of growth expected to take place in sub-Saharan Africa (SSA) and South Asia (SA) (UN-DESA, 2015). These regions also have some of the lowest rates of urban sanitation coverage with only 41% in SSA and 67% in SA estimated to have access to improved sanitation by 2015 (WHO/UNICEF, 2015a). Moreover, up to 90% of residents in cities of low-income countries use on-site sanitation, such as pit-latrines, with minimal options for safe treatment and disposal (Peal et al., 2014).

The Sustainable Development Goal (SDG) sanitation target looks to achieve universal access to sanitation by 2030 (UN, 2015). Notably, the SDG target moves beyond the Millennium Development Goal concept of "improved" sanitation as "access to a toilet facility" and, recognizing the need for safe management of fecal waste along the entire sanitation service chain (SSC), redefines the target indicator as 'safely managed sanitation services' (WHO/UNICEF, 2015b). In line with this expanded scope, a greater proportion of investment will be directed to conveyance, treatment and safe disposal / reuse infrastructure in the coming decades. Moreover, to meet the goal of universal access, major investments will be required for rapidly growing unserved populations.

Increasingly variable climatic conditions, water scarcity, and energy insecurity will further intensify global sanitation service challenges (Howard et al., 2010; Sherpa et al., 2014). Conventional sewerage and wastewater treatment requires substantial natural and financial resources to build and operate, and produces considerable greenhouse gas (GHG) emissions through energy inputs and the decomposition of organic waste (Herzog, 2005; Whittington et al., 2008). Global climate change is projected to decrease water availability in already water stressed regions, while increasing precipitation and flooding risk in wet regions (IPCC, 2014). Both conditions adversely affect sanitation: conventional sewerage and wastewater treatment is prone to operational failure without adequate water and energy, while flooding in densely populated urban slums where on-site sanitation (e.g. pit latrines) are used results in widespread fecal contamination and an associated burden of diarrheal disease.

Conventional sewer networks will not be accessible to the vast majority of SSA's urban residents in the foreseeable future given high capital and operational costs as well as substantial water and energy resource demands, which are often limited and unreliable in these settings (Corominos et al., 2013). Most urban households will instead continue to use on-site sanitation facilities such as pit-latrines, with financing, construction, and maintenance being the responsibility of users themselves (UNICEF/WHO, 2014; Blackett et al., 2014). In recent years, non-sewered and decentralized sewered approaches have received attention as alternatives that can help expand excreta management options for developing country cities. Non-sewered approaches are based on the principle of organizing a system for the "safe and systematic collection, transport, treatment and disposal/reuse of raw or partially digested slurry or solids" (Tilley et al., 2014). Non-sewered systems do not require communal sewer networks, and thus minimize the volume of water required for conveyance. By contrast, decentralized sewered systems move wastes through sewers to local small-scale treatment facilities. As such, they typically require considerably less water and energy inputs as compared to conventional sewerage and wastewater treatment (Gutterer et al., 2009). Moreover, because they generate concentrated waste streams with lower water content, systems can be designed to recover energy, nutrients and water from domestic wastes. While these alternative approaches have traditionally been viewed as temporary or 'stop-gap' solutions, they are increasingly considered to be among the few viable long-term sanitation approaches for cities in low-income regions (Peal et al., 2014).

Given the considerable resource requirements of sanitation infrastructure, coupled with increasingly variable climatic conditions and resource scarcity, it is essential that water and energy requirements are integrated into planning for such investments. At a global level, it is also critical that infrastructure decision-makers seek to minimize carbon emissions. It is likely that no single "best" approach to urban sanitation exists, as costs and benefits of different strategies are strongly shaped by contextual socio-economic, physical and institutional factors. Despite the increasing adoption of non-sewered and decentralized sewered approaches, little is known regarding the long-term environmental or financial costs and benefits of such options.

Environmental impacts of centralized, conventional wastewater treatment plants in high-income countries have been analyzed using life cycle assessment (LCA) methods since at least the mid-1990s (Emmerson et al., 1995, Tillman et al., 1998, Dennison et al., 1998). A handful of LCAs have investigated the environmental impacts of passive, decentralized wastewater treatment, although no studies conducted in SSA were found in the peer-reviewed literature (Dixon et al., 2003; Machado et al., 2007; Fuchs et al., 2011). Two LCA studies were found which investigate environmental impacts of dry, on-site sanitation (Flores et al., 2009; Friedrich et al., 2009). However, no studies were found that address the full lifecycle costs of non-sewered approaches in a low-income urban context. Life cycle cost (LCC) approaches have also been applied to sanitation infrastructure, however available studies are primarily focus on specific technologies (Fonseca et al., 2010). Related literature suggests that downsides to non-sewered approaches may exist. For example, there may be substantial environmental costs of transporting excreta and urine *via* use of fossil-fuel powered equipment rather than water (Tillman et al., 1998; Lundin et al., 2000; Flores et al., 2009). Other literature suggests decentralized approaches may incur additional financial costs associated with poor economies of scale (Whittington et al., 2008).

This study aims to contribute to the knowledge and practice of urban sanitation planning by developing an LCA and LCC based framework that allows decision-makers to evaluate the lifecycle water and energy use, GHG emissions, and financial costs and benefits of sanitation infrastructure alternatives including the entire SSC within the scope of analysis. Secondly, we demonstrate use of this framework, while also generating the first comprehensive analyses of the costs and benefits of non-sewered and sewered systems in urban SSA and SA through a comparative analysis of two operational sewered and non-sewered sanitation systems in Zambia and India. These analyses help elucidate long-term impacts of adopting alternative sanitation approaches, highlight key cost drivers between and within sanitation systems, and provide insights regarding the extent to which resource recovery opportunities influence costs and benefits across varying geographical and social contexts.

4.3 Methods

This study utilizes (1) LCA to evaluate the water use, energy use, and GHG emission impacts – quantified in terms of global warming potential (GWP) – and (2) lifecycle cost assessment (LCC) to analyze the financial costs and benefits of decentralized sewered and non-sewered sanitation approaches. These categories were selected based on their relevance for planning and evaluation of sanitation systems. Water use is critical for the functioning of sewered approaches and may not be necessary for non-sewered systems. This consideration is particularly important for sanitation planners in water stressed environments or in low-income urban areas with low rates of water coverage. Although energy use and GWP are often related for other civil infrastructure, analysis of both categories is important for sanitation systems as organic waste processed during treatment can produce substantial GHG emissions unrelated to energy combustion. This selection of impact categories generally corresponds with other LCAs in the literature investigating sanitation infrastructure.

The functional unit of the analysis, used to compare systems on an equivalent basis, was defined as: management of excreta, urine and wastewater associated with sanitation, *per capita per* year.[37] The sanitation service chain (SSC) from containment through disposal/reuse is included in the LCA scope (Figure 4.1). The infrastructure and use phase are included in the analysis, while the decommissioning phase is excluded.[38] Recovery of biogas energy and water resources (co-products) from the SSC is also

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[37] Management of excreta, urine and wastewater in the present analysis is defined according to the Joint Monitoring Programme (JMP) 'safely managed sanitation' service level, which is defined by the JMP as "use of an improved sanitation facility which is not shared with other households and where excreta are safely disposed in situ or transported and treated off-site" (WHO/UNICEF, 2017). An improved sanitation facility is one which is likely to ensure hygenic separation of human excreta from human contact, including flush or pour-flush toilets to piped sewer systems, septic tanks or pit latrines; ventilated improved pit latrines, pit latrines with a slab, and composting toilets (WHO/UNICEF, 2017). No quantitative criteria for discharge standards of effluent or sludge are specified for 'safely managed services' as defined by the JMP.

[38] Similar LCA studies analyzing wastewater infrastructure have shown the decommissioning phase to have negligible impacts relative to total lifecycle impacts (Emmerson *et al*., 1995; Pillay *et al*., 2004).

considered, including the extent to which reuse of these co-products offsets conventional resource use. Although reuse of nutrients (e.g. nitrogen and phosphorus) from waste is another potential co-product, nutrient costs and benefits are not considered in the present analysis (Asano et al., 2007).

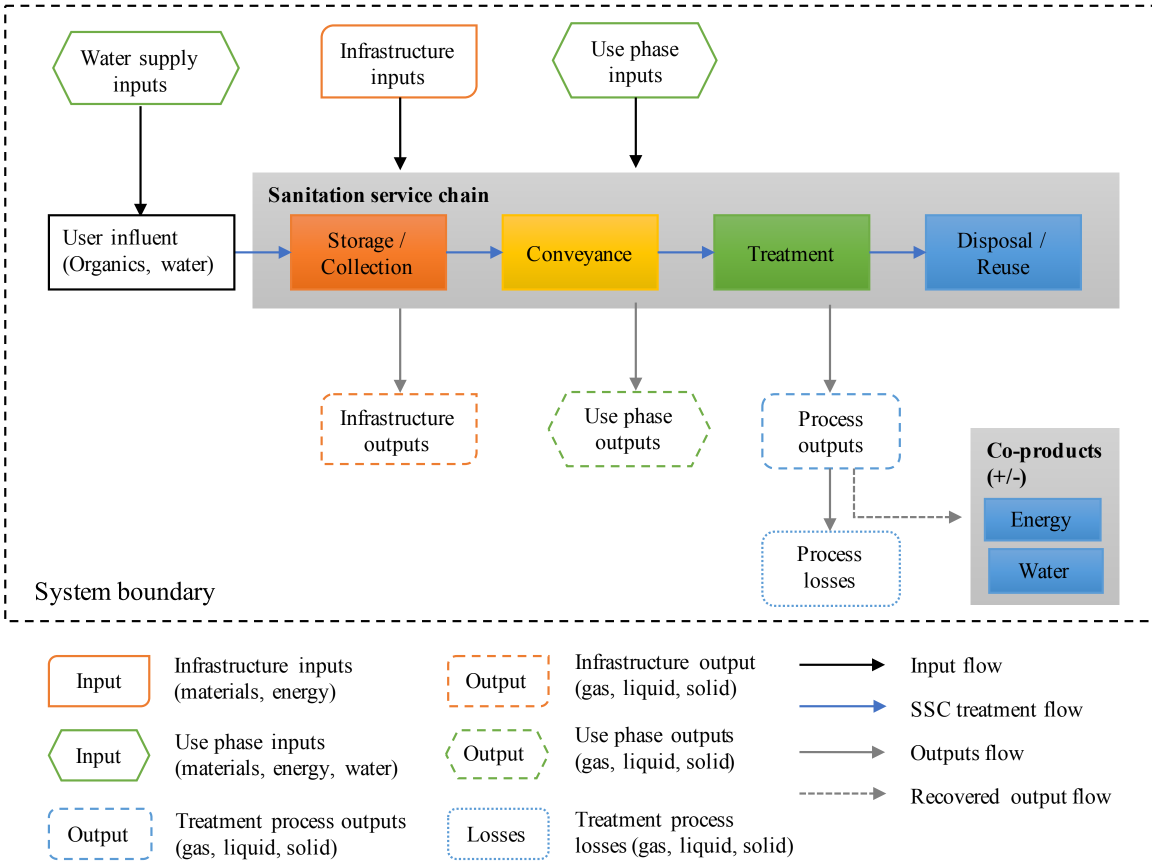

Figure 4.1: Generic lifecycle model and system boundary for sanitation service chain and co-product recovery displaying inputs, outputs and losses

4.3.1 Infrastructure phase

Energy use and GWP for infrastructure materials, construction works and transportation are quantified using SimaPro MultiUser v8.3 software with Eco-Indicator 99 impact assessment methodology. Infrastructure inputs include (1) materials required for construction of sanitation infrastructure based on project bills of quantities and field measurements (e.g. cement, steel), (2) processes required for construction (e.g.

excavation) and (3) transportation of materials from the point of extraction, to manufacturer, supplier and finally to the project site taking into consideration the various vehicle or vessel type required between each step.

To calculate an annual *per-capita* impact for each category, the total energy use and GWP calculated for sanitation infrastructure is then divided by the assumed lifespan (number of years) and number of people served by the sanitation system. A mean lifespan of 20 years is assumed for the main treatment infrastructure and sewer pipelines (Gutterer et al., 2009).[39] Lifespans for other infrastructure and equipment are documented for each case study (see Appendix B.1).

The number of people served for a sewered system is calculated based on the number of households with sewer connections multiplied by the average number of people *per* household. It is often more difficult to estimate the population served by non-sewered systems due to (1) irregular pit emptying intervals, (2) an unknown or inconsistent number of people using a pit latrine between emptying events and (3) incomplete emptying of pits within a community of known size. A method to estimate the number of people served by the non-sewered treatment systems is presented in Section 4.3.2.3.

4.3.2 Use phase

Use phase impacts are modeled in Excel for the defined LCA system boundary (Figure 4.1). Various guidelines, energy and emissions factors, and biochemical processes are used to calculate impacts for (1) water resources, (2) energy resources and (3) GWP as outlined below. Empirical data were collected for each study site *via* on-site measurements, data provision by project operators, project technical reports and published literature containing data specific to the projects. Secondary literature sources are used when site-specific data are not available.

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[39] A 20-year lifespan is less than typically assumed for civil works. However, it is assumed to be appropriate for decentralized sanitation infrastructure in a developing country context. Infrastructure lifespan is modeled in a sensitivity analysis for each system.

4.3.2.1 Water resources (use phase) (m^3/cap/year)

Water resources during the use phase include water used by the population served (user inputs) for operation of the sanitation system, additional water inputs over the SSC and use of water resource co-products (Equation 4.1).

$$W_T = \sum_j W_{U,j} + \sum_j W_{A,j} - \sum_j W_{C,j} \tag{4.1}$$

Where:

W_T	=	total water resources in use phase (m^3/*capita*/yr), from…
$W_{U,j}$	=	…user inputs,
$W_{A,j}$	=	…additional water inputs,
$W_{C,j}$	=	…co-products (water resource offsets *via* resource recovery),
j	=	each treatment system or discharge pathway.

Water resources: user inputs, $W_{U,j}$ (m^3/capita/year)

The assumptions used to calculate water resources required by the user to operate the sanitation system (user inputs, $W_{U,j}$) are based on the type of system. For purposes of calculating water use, a differentiation is made between: (1) sewered (2) non-sewered, waterborne and (3) non-sewered, dry systems. Water use for human consumption, hygiene, cooking and other household purposes is not considered in the model. Additionally, water losses during distribution, which can be significant in developing contexts, are considered in a sensitivity analysis.

Sewered: water resources from user inputs are assumed to be the total wastewater generation *per capita* discharged into the sewer system. wastewater generated may include water used directly for sanitation (water discharged into the user interface to flush contents, i.e. flushwater) and may also include wastewater from other household activities (e.g. bathing, washing dishes, i.e. grey-water). Although grey-water may not be used directly for sanitation, it is included in the calculation of water resource requirements for sewered systems as conveyance is dependent on adequate water flows to ensure self-cleansing of sewer pipes.[40] Site-specific wastewater generation rates are used if known. Otherwise, the International Benchmarking Network for Water and Sanitation Utilities (IBNET, 2016) online database is used for country specific *per-capita* water use. A wastewater generation rate of 50-75% of water use is used to estimate wastewater discharged to the sewer system (WHO/UNEP, 2005).

Non-sewered, waterborne: user input water resources are assumed to be flushwater only. Site-specific *per-capita* flushwater rates are used if known. Otherwise, 4 – 10 L/*capita*/day is used for pour-flush toilets, 12 – 30 L/*capita*/day is used for modern cistern-flush toilets and 20-67 L/*capita*/day is used for older cistern-flush toilets (Mara, 1981; Tilley et al., 2014).[41] If water is used for anal cleansing, an additional 4 – 10 L/*capita*/day may be used (Mara, 1981).

Non-sewered, dry: user input water resources are assumed to be negligible (zero) for dry systems. Water inputs for anal cleansing may be included as above if applicable. Other materials for anal cleansing are not considered.

Water resources: additional water inputs, $W_{A,j}$ (m^3/capita/year)

Additional water inputs at each functional group of the SSC are quantified and included in the model. Water use may include, for example, water used during pit emptying operations or for cleaning equipment after discharging fecal sludge at the treatment facility.

[40] For example, a minimum water use of 60 L/*capita*/day is recommended for adequate hydraulic flows through sewer pipes (Eawag, 2014).

[41] Flushwater used for cistern-flush toilets is dependent on cistern size. Older cistern toilets required 10-20 L/flush (Mara, 1981), whereas more modern toilets are designed for 6-9 L/flush (Tilley et al., 2014). The daily *per-capita* water use assumed for cistern flush toilets is calculated based on *per*-flush and *per-capita* figures reported by Mara (1981) for pour-flush toilets (2-3 L/flush and 4-10 L/*capita*/day).

Water resources: co-products (water resource offsets via resource recovery), $W_{I,j}$ (m^3/capita/year)

Offsets in conventional water use resulting from reuse of water co-products ($W_{C,j}$) are quantified using Equation (4.2). The amount of water recovery (m^3/year) at each functional group is quantified ($W_{R,j}$) and multiplied by the extent to which recovered water *replaces*, rather than only *supplements*, conventional resources as determined through site-specific measurements or judgement ($p_{W\text{-}C,j}$).

$$W_{C,j} = \left[\sum_{j} W_{R,j} \bullet p_{W-C,j}\right] / \, CAP_S \tag{4.2}$$

Where:

$W_{C,j}$	=	water co-products (water resource offsets *via* resource recovery) (m^3/*capita*/yr),
$W_{R,j}$	=	water resources recovered at each functional group (m^3/year),
$p_{W\text{-}C,j}$	=	percentage of water recovered which replaces conventional water resource (%),
CAP_s	=	population served by system,
j	=	each functional group.

4.3.2.2 Energy resources (use phase) (MJ/*capita*/year)

Use phase energy resources included in the LCA model are: electricity inputs, energy required for material use, energy required for water supply, vehicular energy, human energy inputs and interactions with energy and water resource co-products (Equation 4.3).

$$E_T = \sum_{j} E_{E,j} + \sum_{j} E_{M,j} + \sum_{j} E_{W,j} + \sum_{j} E_{V,j} + \sum_{j} E_{H,j} - \sum_{j} E_{C,j} \tag{4.3}$$

where:

E_T = total energy resources in use phase (MJ/*capita*/yr), from…
$E_{E,j}$ = …electricity inputs,
$E_{M,j}$ = …material inputs,
$E_{W,j}$ = …water supply,
$E_{V,j}$ = …vehicular use,
$E_{H,j}$ = …human energy,
$E_{C,j}$ = …co-products (energy or water resource offsets *via* resource recovery),
j = each treatment system or discharge pathway.

Energy resources: electricity inputs, $E_{E,j}$ (MJ/capita/year)

Electricity inputs at each functional group ($E_{E,j}$) including, for example, pumping requirements for conveyance or electricity required for treatment processes, are quantified and converted to MJ/*capita*/year using a standard energy conversion and population served (Equation 4.4):

$$E_{E,j} = \left[\sum_{j} \mathrm{Elec}_j\right] / \mathrm{CAP}_S \tag{4.4}$$

where:

$E_{E,j}$ = energy resources from electricity inputs (MJ/*capita*/yr),
Elec_j = electricity use for each functional group,
CAP_S = population served by system,
j = each functional group.

Energy resources: material inputs, $E_{M,j}$ (MJ/capita/year)

Energy use for material inputs include energy required to (1) replace infrastructure and equipment estimated to have a shorter lifespan than the analysis period (e.g. on-site sanitation, tools, pit emptying equipment, biogas piping and stoves) and (2) for recurrent materials (e.g. cement, cleaning agents, personal protection equipment). Site-specific energy factors and material inputs are reported for each case study (see Appendix B.1).

Energy resources: water supply inputs, $E_{W,j}$ (MJ/capita/year)

Energy use for water supply ($E_{W,j}$) is calculated based on the energy required for both supply and distribution. Water inputs include wastewater generation for the population served plus any additional water inputs along the SSC. To calculate total energy, water use at each functional group is multiplied by a site-specific energy factor for (1) water supply and (2) water distribution (Equation 4.5). If water is supplied by vehicle (tanker truck), the energy for water distribution is calculated based on the volume of water supplied, the distance traveled and an energy factor for the vehicle type. Site-specific energy and emissions factors are reported for each case study (see Appendix B.1).

$$E_{W,j} = \left[\sum_{j} W_j \bullet (EnF_{ws,j} + EnF_{wd,j})\right] / CAP_S \tag{4.5}$$

where:

$E_{W,j}$	=	energy use from water supply inputs (MJ/*capita*/yr),
W_j	=	water supply for each functional group,
EnF_{ws}	=	energy factor for water supply at each functional group,
EnF_{wd}	=	energy factor for water distribution at each functional group,
CAP_S	=	population served by system,
j	=	each functional group.

Energy resources: vehicle use, $E_{V,j}$ (MJ/capita/year)

Energy resources for vehicle use ($E_{V,j}$) include energy required for (1) mechanized conveyance of waste (primarily for non-sewered systems) and other vehicle travel and (2) mechanized pumping for both regular and non-regular desludging events (e.g. removal of sludge from on-site sanitation or treatment infrastructure and discharge into secondary treatment systems). Energy use is calculated for (1) conveyance and vehicle use based on type of vehicle, distance traveled, mean load transported, and energy factors for various sized vehicles; and for (2) pumping based on volume of sludge and an assigned energy factor (Equation 4.6). Site-specific energy factors are reported for each case study (see Appendix B.1).

$$E_{V,j} = \left[\sum_{v,j} (D_{v,j} \bullet L_{v,j} \bullet EnF_{v,j}) \bullet (S_j \bullet EnF_{P,j}) \right] / CAP_S \tag{4.6}$$

where:

$E_{V,j}$	=	energy resources from vehicle use (MJ/*capita*/yr),
$D_{v,j}$	=	distance traveled for each vehicle type (km/year) for each functional group,
$L_{v,j}$	=	mean load (tonnes) for each vehicle type *per* functional group,
$EnF_{v,j}$	=	energy factor for each vehicle type (MJ/tonne-km),
S_j	=	sludge volume mechanically pumped for each functional group (m^3/year)
$EnF_{P,j}$	=	energy factor for pumping of sludge (MJ/m^3 sludge),
CAP_s	=	population served by system,
v	=	each vehicle type,
j	=	each functional group.

Energy resources: human energy inputs, $E_{H,j}$ (MJ/capita/year)

Energy resources for human energy inputs ($E_{H,j}$) are calculated using metabolic equivalents (MET) – the ratio of the work metabolic rate to the resting metabolic rate (Ainsworth et al., 2016). One MET is defined as 1 kcal/kg/hour or the energy cost of sitting quietly for one hour. For each activity performed to provide sanitation services, a MET which closely describes the activity is selected from the Compendium of Physical Activities (Ainsworth et al., 2016). Energy use is then calculated according to the region-specific mean mass of the person(s) performing the activity and number of person-hours spent (Equation 4.7). Energy use is quantified for persons performing manual labor only. MET factors and mean body mass is reported for each case study (see Appendix B.1).

$$E_{H,a,j} = \left[\sum_{a,j} MET_{a,j} \bullet T_{a,j} \bullet m_{P,j} \right] / CAP_S \tag{4.7}$$

where:

$E_{H,a,j}$ = energy resources for human energy inputs (MJ/*capita*/yr),
$MET_{a,j}$ = metabolic equivalent for each activity at each functional group (1 kcal/kg/hour),
$T_{a,j}$ = time spent for each activity for each functional group (person-hours/yr),
$m_{P,j}$ = mean mass of person(s) performing activity (kg),
1/239 = conversion factor: 1 MJ = 239 kcal,
CAP_s = population served by system,
a = each activity type,
j = each functional group.

Energy resources: co-products (energy resource offsets via resource recovery) , $E_{C,j}$ (MJ/capita/year)

Offsets in conventional energy use resulting from reuse of energy or water co-products ($E_{C,j}$) are included in the model. First the amount of energy (MJ/year) and water (m^3/year) recovery at each functional group is quantified. The extent to which energy and water resources *replace*, rather than only *supplement*, conventional resources must be determined through site-specific measurements for existing projects or estimated for future projects. The model additionally considers varying energy efficiencies. For example, Anozie et al. (2007) estimate the cooking energy use efficiency of fuelwood, gas and electricity to be 25%, 73% and 90% respectively.[42] Thus, when considering relative energy efficiencies, the introduction of biogas supply will likely have the highest energy use impact in settings where fuelwood is the primary fuel source and will have the lowest energy use impact where electricity is the primary fuel source. Offsets in conventional energy use resulting from use of co-products are quantified using site and fuel-specific energy factors *per* Equation (4.8) (see Appendix B.1 for energy factors for each site).

$$E_{C,j} = \left[\sum_j E_{R,j} \bullet p_{E-C,j} \bullet \eta_{E-C,j} + \sum_j W_{R,j} \bullet EnF_w \bullet p_{W-C,j} \right] / CAP_S \tag{4.8}$$

[42] Efficiency for charcoal is assumed to be equal to that of fuelwood. Similar efficiencies for charcoal cookstoves are reported by Kaoma and Kasali (1994) in Zambia. The efficiency of biogas is assumed to be equal to the efficiency of propane gas in the Anozie et al. (2007) study.

where:

$E_{C,j}$	=	energy co-products (energy resource offsets *via* resource recovery) (MJ/*capita*/yr),
$E_{R,j}$	=	energy resources recovered at each functional group (MJ/year)
$p_{E\text{-}C,j}$	=	percentage of energy recovered which replaces conventional energy resource (%),
$\eta_{E\text{-}C,j}$	=	relative efficiency of fuel(s) replaced (proportioned *per* fuel type(s) used)
$W_{R,j}$	=	water resources recovered at each functional group (m^3/year)
EnF_w	=	energy factor for water supply (kWh/m3 water) at each functional group
$p_{W\text{-}C,j}$	=	percentage of water recovered which replaces conventional water resource (%),
CAP_s	=	population served by system,
j	=	each functional group.

4.3.2.3 Global warming potential (GWP) (use phase): (kgCO_2e/*capita*/year)

Quantification of GWP (kgCO_2e/*capita*/year) across the SSC considers emissions resulting from electricity, material and water inputs; vehicle use; CH_4 and nitrous oxide (N_2O) emissions produced during biological treatment processes; reduced emissions *via* methane (CH_4) recovery and reduced emissions resulting from replacement of conventional energy and water resources with recovered co-products (Equation 4.9). To allow for summation, all emissions are converted to carbon dioxide equivalents (CO_2e) using 100-yr global warming potential (GWP) factors (IPCC, 2012).

$$C_T = \sum_j C_{E,j} + \sum_j C_{M,j} + \sum_j C_{W,j} + \sum_j C_{V,j} + \sum_j C_{P-CH4,j} + \sum_j C_{P-N2O,j} - \sum_j C_{R,j} - \sum_j C_{C,j} \quad (4.9)$$

where:

C_T	=	total GWP ($kgCO_2e$/*capita*/yr), based on…
$C_{E,j}$	=	…electricity inputs,
$C_{M,j}$	=	…material inputs,
$C_{W,j}$	=	…water supply,
$C_{V,j}$	=	…vehicle use,
$C_{P\text{-}CH4,j}$	=	…treatment process CH_4 emissions,
$C_{P\text{-}N2O,j}$	=	…treatment process N_2O emissions,
$C_{R,j}$	=	…recovered CH_4,
$C_{C,j}$	=	…co-product emission offsets
j	=	each treatment system or discharge pathway.

CO_2e emissions: electricity inputs ($C_{E,j}$, $kgCO_2e$/capita/year)

CO_2e emissions resulting from electricity inputs ($C_{E,j}$) are calculated based on annual electricity use for each functional group and emissions factors for mode of electricity generation specific to the system location (Equation 4.10). Site-specific electricity emission factors are reported for each case study (see Appendix B.1).

$$C_{E,j} = \left[\sum_{j} \mathrm{Elec}_j \bullet \mathrm{EF}_{e,j}\right] / \mathrm{CAP_S} \tag{4.10}$$

where:

$C_{E,j}$	=	CO_2e emissions from electricity inputs ($kgCO_2e$/*capita*/yr),
$Elec_j$	=	electricity use (kWh/year) for each functional group,
EF_e	=	emissions factor for mode of electricity generation ($kgCO_2e$/kWh),
CAP_S	=	population served by system,
j	=	each functional group.

CO_2e emissions: Material inputs ($C_{M,j}$, $kgCO_2e$/capita/year)

CO_2e emissions from material inputs includes the emissions resulting from (1) replacement of infrastructure and equipment estimated to have a shorter lifespan than the analysis period (e.g. on-site sanitation, tools, pit emptying equipment, biogas piping and stoves) and (2) recurrent materials (e.g. cement, cleaning agents, personal protection equipment). Site-specific material inputs and emission factors are reported for each case study (see Appendix B.1).

CO_2e emissions: Water supply inputs ($C_{W,j}$, $kgCO_2e$/capita/year)

CO_2e emissions resulting from water supply inputs ($C_{W,j}$) are calculated based on water use *per* year, site-specific energy factors for water supply and distribution at each functional group and site-specific emissions factor for electricity generation (Equation 4.11). If water is supplied by vehicle (tanker truck), the emissions resulting from water distribution are calculated based on the volume of water supplied, the distance traveled and an emissions factor for the vehicle type. Water use, energy factors and emissions factors are reported for each case study (see Appendix B.1).

$$C_{W,j} = \left[\sum_{j} W_j \bullet (EnF_{ws,j} + EnF_{wd,j}) \bullet EF_e \right] / CAP_S \tag{4.11}$$

where:

$C_{W,j}$	=	CO_2e emissions from water supply ($kgCO_2e$/*capita*/yr),
W_j	=	water supply (m^3/year) for each functional group,
EnF_{ws}	=	energy factor for water supply (kWh/m^3 water) at each functional group,
EnF_{wd}	=	energy factor for water distribution (kWh/m^3 water) at each functional group,
EF_e	=	emissions factor for mode of electricity generation,
CAP_S	=	population served by system,
j	=	each functional group.

CO_2e emissions: Vehicle use ($C_{V,j}$, $kgCO_2e$/capita/year)

CO_2e emissions resulting from vehicle use ($C_{V,j}$) include emissions from (1) mechanized conveyance of waste (primarily for non-sewered systems) and other vehicle travel and (2) mechanized pumping for both regular and non-regular desludging events. CO_2e emissions are calculated for (1) vehicle use based on distance traveled, mean load transported, and emissions factors for types of vehicles; and (2) pumping based on volume of sludge and an assigned emission factor (Equation 4.12). Distances, loads, pumping requirements and emissions factors are reported for each case study (see Appendix B.1).

$$C_{V,v,j} = \left[\sum_{v,j}(D_{v,j} \bullet L_{v,j} \bullet EF_{v,j}) + (S_j \bullet EF_{P,j})\right] / CAP_S \tag{4.12}$$

where:

$C_{V,j}$	=	CO_2e emissions from vehicle use ($kgCO_2e$/*capita*/yr),
$D_{v,j}$	=	distance traveled for each vehicle type for each functional group (km/year),
$L_{v,j}$	=	mean load for each vehicle type for each functional group (tonne),
$EF_{V,j}$	=	emissions factor for each vehicle type ($kgCO_2e$/tkm),
S_j	=	sludge volume mechanically pumped for each functional group (m^3/year)
$EF_{P,j}$	=	emissions factor for pumping of sludge ($kgCO_2e/m^3$ sludge),
CAP_s	=	population served by system,
v	=	each vehicle type,
j	=	each functional group.

CO_2e emissions: Treatment process CH_4 emissions ($C_{P\text{-}CH4,j}$, $kgCO_2e$/capita/year)

The basis for calculating treatment process methane emissions ($C_{P\text{-}CH4,j}$) follows Chapter 6 of the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). The IPCC guidelines calculate CH_4 emissions from wastewater treatment processes as a function of the amount of organic waste generated by a population and an emission factor that characterizes the extent to which a treatment system or discharge pathway is anaerobic *versus* aerobic.

The method presented is adapted from these guidelines to enable calculation of CH_4 emissions over the SSC of a sewered or non-sewered system, which may have several treatment and discharge steps. The concepts presented in the first section are applicable to both sewered and non-sewered systems. However, the computation for each system type is slightly different; for sewered systems the *concentration of organic waste* in wastewater (g/L of COD) is used to enable calculation of gas partitioning. For non-sewered systems the *total organic waste* (g of COD) is used for emission calculations. Additionally, for non-sewered systems with on-site pit latrines as the first step in the SSC, a method to calculate the number of users served by a fecal sludge treatment plant, and the extent of COD degradation during storage in the pit, is presented.

CO_2e: Treatment process CH_4 emissions – Sewered and non-sewered

Treatment process emissions are calculated according to Equation (, adapted from IPCC Equation (6.1):

$$\sum_{j} C_{P-CH4,j} = GWP_{CH4} \bullet \left[\sum_{j} EF_{T-CH4,j} \bullet COD_{rem,j}\right] \tag{4.13}$$

where:

$C_{P\text{-}CH4,j}$	=	CO2e emissions from treatment process CH_4 emissions ($kgCO_2e$/*capita*/yr)
GWP_{CH4}	=	100-yr global warming potential of CH_4 (IPCC, 2012)
$EF_{T\text{-}CH4,j}$	=	emissions factor at each treatment step or discharge pathway (kg CH_4/kg COD)
$COD_{rem,j}$	=	COD removed in each treatment step or discharge pathway (kg COD/*capita*/day)
j	=	each functional group

The emission factor ($EF_{P\text{-}CH4,j}$) at each functional group is a function of the maximum CH_4 producing potential (B_o) and the methane correction factor (MCF) (Equation 4.14) based on IPCC Equation (6.2):

$$EF_{P-CH4,j} = B_o \bullet MCF_j \tag{4.14}$$

where:

$EF_{P\text{-}CH4,j}$	=	emissions factor at each functional group ($kgCH_4$ / kg COD),
B_o	=	maximum CH_4 producing capacity, $kgCH_4$ / kg COD,
MCF_j	=	methane conversion factor at each treatment step or discharge pathway,
j	=	each treatment step or discharge pathway.

B_o is the maximum amount of CH_4 that can be produced from a given quantity of organics (expressed in BOD or COD) (IPCC, 2006). The IPCC guidelines specify a default value of B_o as 0.25 kg CH_4/kg COD, based on biochemical equations to characterize methane at standard temperature and pressure (STP).[43] The IPCC guidelines

[43] Biochemical equations for calculating B_o, the default maximum CH_4 producing capacity: Density of methane at STP: (16.04g CH_4 / 1 mol CH_4) / (22.4L CH_4 / 1 mol CH_4) = **(0.72kg CH_4 / m^3 CH_4)**.

further specify that a COD-based default value of B_o can be converted into a BOD-based value by multiplying by a factor of 2.4.[44]

The MCF is an indication of the extent to which a functional group is anaerobic and therefore the extent to which the CH_4 producing potential (B_o) is realized in each step (IPCC, 2006). Default MCF values are reported in Chapter 6 of IPCC (2006) for various treatment systems and discharge pathways and in Chapter 6 of IPCC (2013) for constructed wetlands (IPCC, 2013). MCFs assumed for each functional group are reported for each case study (see Appendix B.1).

It is assumed that all biodegradable COD will be removed over the SSC, either during treatment processes or at the final discharge pathway. Thus, the initial COD influent, expressed as total organics in wastewater (TOW) is equal to the sum of COD removed (COD_{rem}) at each step of the SSC, including degradation of remaining COD after final discharge Equation 4.15:

$$\mathrm{TOW} = \sum_{j} \mathrm{COD}_{\mathrm{rem,j}} \tag{4.15}$$

where:

TOW	=	total biodegradable organics in wastewater (kg COD/day)
$COD_{rem,j}$	=	COD removed in each treatment or discharge pathway (kg COD /day)

The total biodegradable organics in wastewater (TOW) *per* day may be calculated according to various methods. The simplest method is to estimate TOW based on the number of people served and the organic load generated *per-capita* (Equation 4.16):

$$\mathrm{TOW} = \mathrm{CAP_S} \bullet \mathrm{COD_{cap}} \tag{4.16}$$

Combustion of methane: $CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$. Stoichiometry: (2 mol O_2 / 1 mol CH_4). Multiplying by (32g O_2 / 1 mol O_2 gives (64g O_2 / 1 mol CH_4) or (64g COD / 1 mol CH_4). Therefore, at STP: (22.4L / 1 mol CH_4) / (64g COD / 1 mol CH_4) = **(0.35m^3 CH_4 / kg COD)**. Therefore, B_o at STP: (0.35m^3 CH_4 / kg COD) * (0.72kg CH_4 / m^3 CH_4)) = **(0.25kg CH_4 / kg COD)**.

[44] 0.25 kg CH_4/kg COD * 2.4 = 0.6 kg CH_4/kg BOD

where:

TOW = total biodegradable organics in wastewater (kg COD/day),
CAP_S = population served by sanitation system,
$COD_{b,cap}$ = *per-capita* COD production (g/*capita*/day),

For sewered systems in the present analysis, TOW was calculated using the *per-capita* COD production inferred *via* a mass balance approach (Equation 4.18). The *per-capita* COD production rate calculated for the sewered system is also used to calculate the TOW for non-sewered systems analyzed in the respective countries.

CO_2e: Treatment process CH_4 emissions from sewered systems

COD concentration (mg/L) is used as the basis to calculate CH_4 emissions for sewered systems to enable gas partitioning at varying COD concentrations. Influent COD concentration is calculated according to Equation (4.17):

$$\mathrm{COD_{in}} = \frac{\mathrm{COD_{b,cap}}}{\mathrm{WW_{cap}}} \tag{4.17}$$

where:

COD_{in} = influent COD (mg/L),
COD_{cap} = *per-capita* biodegradable COD production (g/*capita*/day),
WW_{cap} = *per-capita* wastewater generation (L/*capita*/day),

If *per-capita* COD is not known, influent COD may be inferred using a mass (Equation 4.18) (Laramee et al., 2017). This approach requires collection of biogas production, wastewater flow and site data. Where a system is operational and this information is possible to collect, a mass balance calculation can be more accurate than relying on approximations for *per-capita* COD production, for which little data are available for low-income populations. This mass balance approach may also be used to calculate influent COD at any stage of the treatment system where biogas production and wastewater flow can be measured.

$$COD_{in} = COD_{out} + \frac{CH_{4\,(g)} \bullet f_{bg}}{Q} + \frac{CH_{4\,(l)} \bullet f_{bg}}{Q} \tag{4.18}$$

where:

COD_{in}	=	influent COD (mg/L),
COD_{out}	=	effluent COD (mg/L),
$CH_{4(g)}$	=	CH_4 released in the gas phase (m^3/day),
$CH_{4(l)}$	=	CH_4 that remains dissolved in the liquid effluent (m^3/day),
Q	=	wastewater flow-rate (m^3/day),
f_{bg}	=	conversion factor of CH_4 to COD adjusted for site conditions.[45,46]

Distribution of biogas between methane (CH_4), carbon dioxide (CO_2), and nitrogen (N_2), is calculated as a function of influent COD concentration and wastewater temperature, following theoretical equilibrium methods described by Cakir and Stenstrom (2005) and McCarty et al. (2016). The percentage of produced CH_4 in the gas phase *versus* dissolved in the liquid phase is also calculated using these equilibrium methods. For the purposes of this model, it is assumed that CH_4 in the gas phase is potentially recoverable in simple, decentralized systems, whereas dissolved CH_4 is not recovered and instead is released upon exiting the treatment system. Estimation of dissolved CH_4 is particularly relevant for wastewater with low organic strengths.

If the mass balance is used to calculate influent COD concentration, an iterative method is required with linear interpolation between incremental COD values to infer the fraction of CH_4 in the gas and liquid phases. $CH_{4(g)}$ production (m^3/day) is then computed using measured biogas production and the theoretical equilibrium CH_4 content in the gas phase.

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45 Soto et al. (1993) report a factor f_{bg} = (1/385) g COD ml CH4-1 at 20°C. This factor may be adjusted to site conditions (temperature, elevation) using the Ideal Gas Law.

46 The mass balance equation assumes that measured COD in the effluent is entirely biodegradable, and that COD removed in settled sludge is negligible. While there is some accumulation of COD in an anaerobic reactor, the long residence time (3-5 years) of the sludge allows for near-complete degradation of the biodegradable COD. Thus, the sludge accumulation rates in the BGDs are principally due to accumulation of inert solids and non-biodegradable COD, which are neglected in this analysis. The mass balance approach therefore underestimates the total influent COD concentration, and thus the calculated *per-capita* COD production and treatment efficiency as well.

Finally, $CH_{4(l)}$ (m^3/day) production is calculated using the theoretical equilibrium CH_4 content in the liquid phase and $CH_{4(g)}$ production (Equation 4.19):

$$CH_{4(l)} = CH_{4(l)}(\%) * \frac{CH_{4\,(g)}}{1 - CH_{4(l)}(\%)} \tag{4.19}$$

where:

$CH_{4(g)}$ = CH_4 released in the gas phase (m^3/day),
$CH_{4(l)}$ = CH_4 that remains dissolved in the liquid effluent (m^3/day unless noted in equation as %).

After calculating influent COD to the SSC, the COD removed at each treatment step or discharge pathway is calculated using measured data for existing systems or design assumptions for planned systems. An MCF is assigned to each treatment step, with CH_4 emissions calculated *per* Equations (4.13) to (4.15).

CO_2e: Treatment process CH_4 emissions from non-sewered systems

The total COD load, rather than COD concentration, is used to calculate CH_4 emissions from non-sewered systems. All CH_4 produced is assumed to be released in the gas phase.[47] Equation (4.16) is used to calculate the total COD load *per* day, based on the *per-capita* biodegradable COD production rate and the total population served by the system. Population-specific or default IPCC values may be used *for per-capita* COD production. Typically, the data required to calculate *per-capita* COD production using a mass balance (e.g. Equation 4.18) will not be available for conventional pit latrines with long time intervals between emptying events as substantial degradation will have occurred during storage.

If the population served by a fecal sludge treatment plant is unknown, an estimate of the population is computed based on the mean total solids input into the system *per*-day, literature estimates for daily *per-capita* feces production, and estimated degradation of

[47] Theoretical equilibrium equations show dissolved CH_4 at approximately 1% for COD strengths above 5,000 mg/L. COD concentration of fecal sludge is reported in the range of 20,000 – 50,000 mg/L for public toilets (assumed to be non-flush) and 1,200 – 10,000 from septic tanks (Niwagaba et al., 2014).

solids during storage in pit latrines (Equation 4.20). A similar approach is also described to calculate COD reduction during storage in a pit latrine.

$$CAP_S = \frac{F_{TS,dry} \bullet 1000}{f_{d,dry}} \tag{4.20}$$

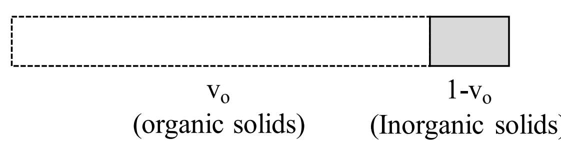
where:

CAP_S = population served by system,
$F_{TS,dry}$ = total solids fecal sludge input into system (kg/day, dry mass),
$f_{d,dry}$ = feces produced *per capita per* day at depth "d" adjusted for degradation during storage in pit (g/*capita*/day)

The mean dry solids produced *per capita per* day in low-income countries is estimated at 39 g (+/- 14.1 g) (Rose et al., 2016).[48] This figure, which represents the dry mass of fresh feces, is adjusted according to the estimated extent of degradation during storage using the percentage of organic solids measured at various pit latrine depths reported by Nwaneri et al. (2008). The following method is used to account for degradation of organics during pit latrine storage:

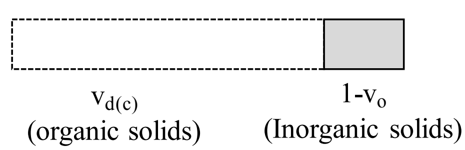
1 gram (g) of fresh feces contains:

- v_o (g) volatile organic solids and
- $1-v_o$ (g) inorganic solids.



After degradation, the original 1g sample of fresh feces theoretically contains an identical inorganic solids content ($1-v_o$) (no change from degradation processes) and a reduced, or 'corrected' volatile organic solids content ($v_{d(c)}$):

- $v_{d(c)}$ (g) volatile organic solids and
- $1-v_o$ (g) inorganic solids.



[48] The range of mean dry solids reported by Rose et al., (2016) was examined in a sensitivity analysis for each system.

The percentage of volatile organic solids measured at depth "d", $v_{d(m)}$ is:

$$v_{d(m)} = \frac{v_{d(c)}}{v_{d(c)} + (1 - v_o)} \tag{4.21}$$

Solving for $v_{d(c)}$ (the 'corrected' volatile organic solids content):

$$v_{d(c)} = \frac{v_{d(m)} \bullet (1 - v_o)}{(1 - v_{d(m)})} \tag{4.22}$$

The daily *per-capita* feces production is then adjusted from the estimated fresh feces dry weight ($f_{o(dry)}$) to an after degradation dry weight at depth "d" ($f_{d(dry)}$):

$$f_{d(dry)} = f_{o(dry)} \bullet \frac{(v_{d(c)} + (1 - v_o))}{1} \tag{4.23}$$

Example 4.1:

The following example demonstrates the procedure used to calculate the corrected organic solids content and adjusted *per-capita* feces production. Nwaneri et al. (2008) report an organic solids content of fresh feces as 0.84 g organic solids/g dry sample and an inorganic content of 0.16 g inorganic solids/g dry sample. At a pit latrine depth of 0.5 m, the mean measured organic solids content is reported as 0.48 g organic solids/g dry sample (Nwaneri et al., 2008). Using Equation (4.22), the "corrected" organic volatile solids content at depth d = 0.5 m ($v_{0.5(c)}$) is:

$$v_{0.5(c)} = \frac{0.48\text{g} \bullet (1 - 0.84\text{g})}{(1 - 0.48\text{g})} = \frac{0.22 \text{ g organic solids}}{\text{g dry sample}}$$

The daily *per-capita* feces production is then adjusted from the estimated fresh feces dry weight ($f_{o(dry)}$) to an after degradation dry weight at depth, d = 0.5 m ($f_{0.5(dry)}$) according to Equation (4.23):

$$f_{0.5(dry)} = \frac{39\text{g}}{capita \bullet \text{day}} \bullet \frac{0.15\text{g} + (1 - 0.84\text{g})}{1\text{g}} = \frac{12\text{g}}{capita \bullet \text{day}}$$

Following a similar procedure, the corrected organic solids content and adjusted daily *per-capita* feces production after degradation are reported in

Table 4.1 for other depths measured by Nwaneri et al. (2008).

Table 4.1: Corrected solids content and daily *per-capita* feces production after degradation in pit latrine

Layer	Mean measured organic solids ($v_{d(m)}$) (g organic solids / g dry sample)[a]	Corrected organic solids at depth, d ($v_{d(c)}$) (g organic solids / g dry sample)[b]	Adjusted daily *per-capita* feces production ($f_{d,\,dry}$) (g/cap/day)[c]
Fresh feces	0.84	0.84	39
Surface layer	0.58	0.22	15
Depth = 0.5 m	0.48	0.15	12
Depth = 1 m	0.34	0.08	9

[a] Values reported by Nwaneri et al., 2008
[b] Calculated value (Equation (4.22))
[c] Calculated value (Equation (4.23))

The mean adjusted daily *per-capita* feces production for a specific depth emptied is then calculated as the average of values from the surface layer to the depth emptied.[49] For example, for 1 m depth of fecal sludge removed during a pit emptying event, the mean daily *per-capita* production of feces after degradation for the entire 1m pit depth emptied is calculated as 12 g/cap/day.

A similar method is used to calculate the extent of COD degradation during storage in a pit latrine. Following Equation (4.23) an adjusted COD value is calculated at each depth (COD_d) using measured COD values reported by Nwaneri et al. (2008) (COD_m). The ratio of 'corrected' organic solids calculated ($v_{d(c)}$) and inorganic solids ($1-v_o$) over the original 1 g sample is used to adjust the COD measured at each depth (Equation 4.24):

[49] The calculation for mean adjusted daily *per-capita* feces production assumes the amount of fresh feces collected during the pit emptying event is negligible as the time between emptying is relatively long (over 1 year) and Nwaneri et al. (2008) hypothesize that degradation of top layer occurs relatively quickly.

$$\mathrm{COD_d} = \mathrm{COD_m} \bullet \frac{(\mathrm{v_{d(c)}} + (1 - \mathrm{v_o}))}{1} \tag{4.24}$$

The percent of COD degradation at each depth is then calculated as a percent reduction from the measured fresh feces COD (Table 4.2). The mean adjusted COD degradation for a specific depth emptied is calculated as the average of values from the surface layer to the depth emptied. CH_4 emissions during pit latrine storage are then calculated using Equation (4.13) to (4.14) according to the COD removed during containment and an assumed MCF based on the extent of anaerobic conditions in the pit (Table 6.3, IPCC 2006). CH_4 emissions from the remaining COD after pit containment is calculated according to the MCFs and % COD removed in the subsequent treatment systems and discharge pathways along the SSC following Equations (4.14) to (4.15).

Table 4.2: Adjusted COD and percent COD degradation at various depths after storage in pit latrine

Layer or depth (m)	Mean measured COD ($COD_{d(m)}$) (mg COD / g dry sample)[a]	Corrected COD at depth, d (COD_d) (mg COD / g dry sample)[b]	Total COD degradation at depth, d (%)[c]
Fresh feces	1110	1110	0%
Surface layer	540	206	81%
Depth = 0.5 m	380	117	89%
Depth = 1 m	280	68	94%

[a] Values reported by Nwaneri et al., 2008
[b] Calculated value (Equation (4.23))
[c] Calculated value (Equation (4.24))

CO_2e emissions: Treatment process N_2O emissions ($C_{P\text{-}N2O,j}$, $kgCO_2e$/capita/year)

The basis for calculating treatment process nitrous oxide (N_2O) emissions ($C_{P\text{-}N2O,j}$) follows IPCC (2006) Section 6.3, the IPCC (2013) supplement for constructed wetlands for waterborne systems, and IPCC (2006) Section 11.2 for dry systems with land application of sludge. The IPCC guidelines calculate N_2O emissions according to the amount of nitrogen discharged in wastewater multiplied by an emission factor, expressed in terms of kg N_2O-N/kg N. The emission factors are based on literature values and are influenced by the extent of nitrification and denitrification, coverage of vegetation and climatic conditions (IPCC, 2013).

As with CH_4 emissions, the method presented here for N_2O emissions is adapted to evaluate specific treatment systems (or combinations of systems). CO_2e emissions resulting from treatment process, discharge or land application are calculated according to Equation (4.25), based off IPCC Equation (6.7):

$$\sum_j C_{P-N2O,j} = GWP_{N2O} \bullet \left[\sum_j EF_{T-N2O,j} \bullet N_{rem,j}\right] \tag{4.25}$$

where:

$C_{P\text{-}N2O,j}$	=	CO_2e emissions from treatment process N_2O emissions (kgCO_2e/*capita*/year)
GWP_{N2O}	=	100-yr global warming potential of N_2O (IPCC, 2012)
$EF_{T\text{-}N2O,j}$	=	emissions factor at each treatment step or discharge pathway (kg N_2O-N/kg N)
$N_{rem,j}$	=	N removed in each treatment step or discharge pathway (kg N/day)
j	=	each functional group

It is assumed that all nitrogen will be removed over the SSC, either during treatment processes or at the final discharge pathway. Thus, the total nitrogen influent (TN) is equal to the sum of nitrogen removed (N_{rem}) at each step of the SSC and remaining N after final discharge (Equation 4.26):

$$TN = \sum_j N_{rem,j} \tag{4.26}$$

where:

TN	=	total nitrogen in wastewater (kg N/day)
$N_{rem,j}$	=	COD removed in each treatment step or discharge pathway (kg N/day)

The total amount of nitrogen (TN) entering the SSC *per*-day is calculated based on *per-capita* nitrogen generation for the population served using Equation (4.27):

$$TN = CAP_S \bullet N_{cap} \tag{4.27}$$

where:

TN	=	total nitrogen in wastewater (kg N/day)
CAP_S	=	population served by sanitation system
N_{cap}	=	population-specific *per-capita* nitrogen generation (g/*capita*/day)

The population-specific *per-capita* nitrogen generation (N_{cap}) may be estimated based on site-specific data, calculated using field measurements for existing systems or estimated according to *per-capita* protein consumption for the population served multiplied by a factor for the fraction of nitrogen in protein (F_{NPR} = 0.16 kg N/kg protein) (IPCC, 2006).

CO_2e emissions: Recovered CH_4 emissions ($C_{R,j}$, kgCO_2e/capita/year)

$C_{R,j}$ refers to CH_4 emissions recovered from treatment processes, which are combusted (either through energy reuse or flaring) and converted to CO_2.[50] The resulting CO_2 emissions are considered to be biogenic and thus are not considered in the summation of GWP impacts (IPCC, 2006; US-EPA, 2011). All recovered CH_4 emissions are therefore subtracted from the CO_2e emissions balance in Equation 4.9.

CO_2e emissions: co-product emission offsets (emission offsets via energy or water resource recovery) ($C_{C,j}$, kgCO_2e/capita/year)

Co-product emission offsets ($C_{C,j}$) refer to reductions in emissions resulting from replacement of conventional resources with energy and water co-products. The extent of emission offsets depends on the type and efficiency of conventional fuel(s) replaced, the source and amount of energy used for conventional water supply and the extent to which recovered resources *replace* rather than *supplement* conventional resource use (Equation 4.28).[51]

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[50] Combustion of CH_4 produces a molecularly equivalent amount of CO_2 according to the formula:
$CH_4 + 2\ O_2 \rightarrow CO_2 + 2\ H_2O$

[51] Accounting of emissions resulting from biomass fuel use is also dependent on the consumption and regeneration cycle of the specific biomass under consideration (US-EPA, 2011). Typical practices for charcoal production in eastern and southern Africa reportedly result in the felling

Emission reductions *via* energy replacement are calculated by first quantifying the total energy recovery *per* year ($E_{R,j}$ in MJ/year) at each functional group. Recovered energy is then converted to equivalent units of conventional fuel using the site-specific proportions of fuel type(s) replaced (p_e), the energy content for each fuel type (EC_e) and the assumed efficiency of each fuel type. The equivalent units of fuel are then converted to CO_2e emissions using emissions factors (EF_e) for each fuel type and 100-yr global warming potentials. Water reuse ($W_{R,j}$ in m^3/year) is converted to energy resources using site-specific energy factors (EnF_w) and then to CO_2e emissions using emissions factors (EF_e) for mode of electricity generation used for water supply. Fuels used *per* study site, energy contents, energy factors and emissions factors are reported for each case study (see Appendix B.1).

Studies on, for example, fuel efficiency, have demonstrated that improving the efficiency of a product can lead to increased use and thus may not lead to the full reduction in fuel savings expected, a phenomena known as a "rebound" effect. This established phenomena has been well studied, although the magnitude of effect is uncertain, with estimates ranging from 5% to 30% (Gillingham et al., 2013). Therefore, the extent to which energy and water resources *replace* ($p_{E\text{-}C,j}$ and $p_{E\text{-}C,j}$, respectively), rather than only *supplement*, conventional resources needs to be determined through site-specific measurements or, where no data exists, estimated using literature sources derived from similar contexts. In the present analysis, recovered energy is assumed to replace an equivalent amount of conventional energy (*per* MJ), based on findings from Laramee et al. (2017), which suggest that in the context of the systems studied, recovered biogas replaced conventional fuel use, rather than supplemented it, even when biogas was supplied at no cost.

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of entire trees and clear-cutting around kiln sites leading to transformation of vegetation structure and composition over large harvested areas (Chidumayo and Gumbo, 2012). Thus, in this study, we assume household charcoal use results in a long-term decline in the total carbon embodied in standing biomass and a net release of carbon.

$$C_{C,j} = \left[\sum_{e,j} E_{R,j} \bullet p_{e,j} \bullet EC_e \bullet EF_e \bullet p_{E-C,j} \bullet \eta_{E-C,j} + \sum_{j} W_{R,j} \bullet EnF_w \bullet EF_e \bullet p_{W-C,j} \right] / CAP_S \quad (4.28)$$

where:

$C_{C,j}$	=	CO_2e emissions: co-product emission offsets (emission offsets *via* energy or water resource recovery) (kgCO_2e/*capita*/yr)
$E_{R,j}$	=	Energy resources recovered at each functional group (MJ/year)
$p_{e,j}$	=	Percentage of each energy type replaced at each functional group (%)
EC_e	=	Energy content of each energy type (MJ/unit of fuel)
EF_e	=	Emissions factor for energy type (kgCO_2e/unit of fuel)
$p_{E-C,j}$	=	Percentage of energy recovered which replaces conventional energy resource (%)
$\eta_{E-C,j}$	=	relative efficiency of fuel(s) replaced (proportioned *per* fuel type(s) used)
$W_{R,j}$	=	Water resources recovered at each functional group (m^3/year)
EnF_w	=	Energy factor for water supply (kWh/m^3 water)
$p_{W-C,j}$	=	Percentage of water recovered which replaces conventional water resource (%)
CAP_s	=	Population served by system
j	=	each functional group

4.3.3 Economic analysis (US $/*capita*/year)

4.3.3.1 Overview

The *per-capita* equivalent annual cost (EAC) for the user of the system and for the operating agency are quantified using Equation 4.29 to 4.31.

$$\mathrm{EAC} = \frac{NPV}{A_{t,r}}, \quad (4.29)$$

$$A_{t,r} = \frac{1 - \frac{1}{(1+r)^t}}{r}, \quad (4.30)$$

$$\mathrm{NPV}\ (\mathrm{i, N}) = \sum_{t=0}^{N} \frac{R_t}{(1+i)^t} \quad (4.31)$$

where:

A_{tr} = Present value annuity factor
NPV = Net present value
r = Discount rate
t = Number of years
N = Number of periods

To allow for comparison across projects, all costs are converted to US $ in the year 2015. First, the local currency gross domestic product (GDP) deflator is used to convert costs to the year 2015 (WASHCost, 2011; World Bank, 2016a):

$$\text{Local currency}_{2015} = \text{Local currency}_{\text{Yr X}} * \frac{\text{GDP deflator}_{\text{Yr 2015}}}{\text{GDP deflator}_{\text{Yr X}}}. \tag{4.32}$$

Second, the local currency cost in the year 2015 is converted to US $ using the purchasing power parity (PPP) conversion factor (WASHCost, 2011; World Bank, 2016a):

$$\text{US \$}_{2015} = \frac{\text{Local currency}_{2015}}{\text{PPP conersion factor}_{\text{LC 2015 } per \text{ international \$}}} \tag{4.33}$$

Future costs are discounted to the year 2015 using a real discount rate of 5% for local currency financial flows and a real discount rate of 1.2% for US $ financial flows (World Bank, 2013; World Bank, 2016; OMB, 2015).

Costs of each system for the (1) user and (2) operating agency (sanitation service provider) are quantified separately. Costs considered include: (1) capital expenditures (CapEx), operational expenditures (OpEx), and capital maintenance expenditures (CapManEx) (WASHCost, 2011). Further details of these cost categories, as well as allocation of costs to user or agency within these categories, are elaborated in the following sections.

4.3.3.2 Capital expenditure (CapEx)

CapEx for sanitation infrastructure is calculated using itemized construction material quantities and costs provided by the technical consultants of each case study analyzed

(BORDA, 2016; CDD, 2016). All infrastructure from collection/storage to disposal/reuse functional groups are included in the analysis. The superstructure (e.g. latrine walls above ground and roofing) and user interface (e.g. toilet pedestal) costs are excluded from the collection/storage functional group costs (e.g. only latrine substructure and slab is included). If only a material cost list is available, the additional costs for construction (e.g. transportation of materials, labor and supervision) are estimated as 50% of material costs (BORDA, 2016). Additionally, the cost of project management and technical design are estimated to be 25% and 5% of the material plus construction costs, respectively (BORDA, 2016). The additional project management and technical design costs are excluded for household pit latrines. The total infrastructure CapEx is calculated as the sum of material costs, construction costs, project management and technical design. The lifespan for infrastructure is assumed to be 20 years unless noted otherwise in the CapManEx section (Section 4.3.3.3).

For non-sewered systems, it is assumed that collection/storage infrastructure (e.g pit latrines or septic tanks) is built by the household owner or landlord progressively over time rather than at sanitation system implementation (i.e. Year 0). Thus, the total CapEx of collection/storage infrastructure for the population served is assumed to be equally distributed over the lifespan of the collection/storage infrastructure. If the lifespan of the collection/storage infrastructure is less than the analysis period, the cost of replacement is assumed as CapManEx and is also equally distributed over the remainder of the analysis period. The annual *per-capita* cost of collection/storage infrastructure is calculated as:

$$\mathrm{CS}_{\mathrm{cap-yr}} = \frac{\text{Cost } per \text{ CS unit}}{\text{Number of people sharing CS} \bullet \text{Lifespan of CS (years)}} \tag{4.34}$$

Where:

$\mathrm{CS}_{\text{cap-yr}}$ = Cost of collection/storage infrastructure *per capita per* year

Allocation of CapEx to user and agency:

CapEx allocated to the user includes the cost of the household sewer connection to the main sewer line for sewered systems and the cost of collection/storage (e.g. cost of the pit latrine sub-structure) for non-sewered systems. CapEx allocated to the agency includes cost for conveyance infrastructure (e.g. sewer pipes, emptying equipment or vehicle transport), treatment infrastructure and cost of either disposal or resource recovery infrastructure (including biogas piping and stoves) (Table 4.3).

Costs excluded from CapEx:

The following costs are excluded from CapEx for each system: (1) "soft" costs such as community engagement (due to insufficient data); (2) cost of household toilet or superstructure (outside scope of study); (3) household 'coping costs' – i.e. the cost of additional investments such as water storage tank, water filter – absorbed by the household to attain a satisfactory level of service (due to insufficient data); (4) water supply infrastructure (although water is required for sewered systems, it is assumed that water supply infrastructure is primarily built for basic human consumption); (5) road network (although required for conveyance of waste for non-sewered systems where either motorized or human powered conveyance is used, it is assumed that the road network is primarily required for other transportation needs) and (6) secondary site infrastructure (i.e. site walls, office buildings, caretakers building) – if applicable, the cost of site ramps or platforms required for offloading fecal sludge at treatment plants is included (WASHCost, 2011).

Operational expenditures (OpEx)

Operational expenditure (OpEx) includes 'recurrent (regular, ongoing) expenditure on labor, chemicals, materials and purchases of any bulk water; any recurrent maintenance to keep systems running at design performance, but does not include major repairs or renewals' (WASHCost, 2011). OpEx is calculated using data provided by system operators and technical advisors to the projects including BORDA (2016), CDD (2016), KWT (2016), NWWSSC (2016), and WSUP (2016). Secondary literature sources are used when necessary. Data and respective references are reported for each case study (see Appendix B.1).

Labor, material and energy costs

OpEx includes labor costs (caretaker, lab technicians, engineer, administrators, managers), material costs (personal protection equipment, tools, supplies required for recurrent repairs – e.g. cement, chemicals, lab supplies, cleaning supplies) and energy costs (petrol or diesel, electricity). All recurrent labor, material and energy costs required for treatment system operations and maintenance are allocated to the agency (e.g. the water and sewerage utility responsible for operating the sanitation system) (Table 4.3).

Water supply and sanitation surcharge

The cost of water supply for sewered systems is calculated according the volume of wastewater generated and the price *per* volume of water. The volume of wastewater, rather than total water supply, is used for calculation. This is because the difference between water use and wastewater generation is considered as water required for basic consumption rather than for sanitation. For sewered systems, daily wastewater flow was measured; for non-sewered systems assumptions outlined in Section 4.3.2.1 are used to estimate sanitation related water use. In the case of an increasing block tariff, the second-level tariff and higher tariffs, if applicable, are used to calculate the cost of water supply as the lowest tariff is assumed to be for basic human consumption.

The price of water supply is allocated to the user as a cost (Table 4.3). If the agency is also the water service provider, the price of water is assumed to equal the agency's cost of supplying water (IBNET, 2015). Thus, the price of water supply is allocated as both a cost and benefit and is effectively not considered as either a positive or negative financial flow. The water supply cost or benefit to water service providers, other than the user or agency, is not considered in the analysis.

The cost of the sanitation service is calculated as the user fee (e.g. as a flat rate or sanitation surcharge) for sewered systems and as the cost of pit emptying for non-sewered systems. If a sanitation surcharge is used, the total water supply *per* connection is used to calculate the surcharge amount (*versus* the amount of wastewater generated). If the total water supply to each household is not known, a range of 50-75% wastewater generation (from total water supply) is used to estimate the average water *per* household (WHO/UNEP, 1997). If a fixed charge for the water and sewerage connection is used, the ratio of wastewater generation to total water supply is used to estimate the amount of the fixed charge which is allocated to sanitation. The sanitation fee is allocated to the user as a cost and to the agency as a benefit (Table 4.3).

Energy, water and biosolids recovery

For all systems analyzed, biogas was provided free of charge at the time of the study. Therefore, the financial benefit associated with biogas recovery is calculated by multiplying the mean biogas recovered *per* day (m^3/day) by the energy content of biogas (MJ/m^3); the total energy content recovered *per* day (MJ/day) was then converted into an equivalent cost of conventional fuel using relative fuel efficiencies and the energy content and price of conventional fuel(s) used. This calculation assumes the price of conventional fuels are a reasonable proxy for the value of biogas energy. The financial benefit of energy recovery is allocated to the operating agency as a theoretical benefit. Reuse of bio-solids as an energy source is not considered in the present analysis.

When applicable, the financial benefits associated with water recovery and biosolids are assessed similarly to energy recovery. If wastewater offset the use of conventional water supply, the financial benefit is calculated according to the price of conventional water supply and the volume of water offset. Similarly, if reuse of bio-solids offsets soil amendments or fertilizers otherwise used, the financial benefit is calculated according to the price of alternatively used products and the amount of product offset. If the volume of bio-solids produced at the fecal sludge treatment plant is unknown, the amount (kg) of dry bio-solids assumed to be produced *per* day is calculated based on the mean mass of fecal sludge entering the treatment plant multiplied by the mean total solids of fecal sludge into the system (%). This assumes that (1) that the dry weight of fecal sludge entering the system is equal to the dry weight exiting the system and (2) there is no moisture content in the dried sludge. The first assumption likely overestimates the bio-solids produced as some degradation of solids takes place in the treatment plant. However, most degradation is assumed to have occurred during long periods of storage in pit latrines or septic tanks. Conversely, the second assumption likely underestimates the amount (kg) of bio-solids as some moisture will be retained in the dried sludge. The financial benefit of water recovery and bio-solids recovery is allocated to the operating agency either as a real or theoretical benefit.

Carbon pricing

The annual reduction in CO_2e emissions *per capita* is assumed to be the difference in emissions under a "waste(water) treatment" only scenario *versus* a "waste(water) treatment with energy recovery" scenario. The total CO_2e emissions for the waste(water) treatment only scenario is calculated using Equation (4.9) excluding the terms 'recovered CH_4' and 'co-product emission offsets' while the emissions resulting from the waste(water) treatment with energy recovery scenario includes these final two terms. In other words, the difference in CO_2e emissions between treatment scenarios is assumed to be the sum of emissions reduced *via* CH_4 recovery and conventional energy or water use offset. This assumes waste treatment processes, including energy and water use, methane emissions, material inputs, are identical in either scenario other than the recovery of CH_4 and the replacement of either conventional energy or water with recovered resources.

The theoretical revenue which could be accessed *via* carbon offsets is not included in the baseline cost calculation as no system analyzed currently obtains revenue through carbon financing. However, to explore the extent to which carbon pricing could potentially offset costs, the theoretical revenue from carbon offset is calculated under three carbon price scenarios:

- Scenario (1): \$2.40/tCO_2e (based on a recently auctioned price of carbon under the Pilot Auction Facility for Methane and Climate Change Mitigation (PAF) mechanism) (Kossoy et al., 2015).

- Scenario (2): \$37/tCO_2e (equal to the 'social cost of carbon' (SCC) used by the United States government for cost-benefit analyses) (IWG, 2016).[52]

- Scenario (3): \$100/tCO_2e (the minimum carbon price recommended by the United Nations Global Compact) (World Bank and ECOFYS, 2016).

As the potential revenue *via* carbon financing is based on US \$ financial flows (rather than the local currency), all future costs are analyzed in US \$ and discounted to 2015 using the assumed discount rate for US \$. The NPV of carbon financing is added to the NPV of local currency financial flows after it is discounted to 2015 and converted to US \$. The cost for verification of carbon reductions and administration of carbon finance is not considered in the analysis. The potential revenue from carbon financing is allocated to the agency as a benefit (Table 4.3).

Costs excluded from OpEx:

The following costs are excluded from OpEx estimates for all systems: (1) 'soft' costs such as program support and community outreach (due to insufficient data); (2) household coping costs (due to insufficient data); (3) household cleaning supplies or maintenance of user interface (outside scope of study); and (4) overhead costs – e.g. equipment or electricity to operate utility office, vehicles not exclusively used for project (insufficient data).

4.3.3.3 Capital maintenance expenditures (CapManEx)

CapManEx is 'expenditure on asset renewal, replacement and rehabilitation' and includes all major, non-recurrent maintenance events (WASHCost, 2011). Costs and timeframes for CapManEx were provided by project operators, either as actual or estimated figures, and using secondary literature sources (see case study documentation).

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[52] The SCC is an estimate of the economic damages associated with an incremental increase of one metric ton of CO_2 in a given year, or as the value of damages avoided for the reduction of one metric ton of CO_2 (IWG, 2015).

Replacement of on-site sanitation, biogas infrastructure and pit-emptying tools and carts are considered as CapManEx in this study. Lifespans reported in the literature for on-site sanitation (e.g. pit latrines) vary widely and are dependent on materials and construction quality. For the present study, a mean lifespan of 14 years is assumed for pit latrines (Hutton and Haller, 2004; WRC, 2007; Klutse et al., 2010; Tilley et al., 2014; Hutton and Varughese, 2016).[53] After initial construction (which is included in CapEx), replacement of on-site sanitation is assumed to occur progressively over time with the cost of replacement equally distributed over the remainder of the analysis period. The annual *per-capita* cost for replacement is calculated *per* Equation (4.34). A lifespan of 6 years is assumed for gas piping infrastructure and appliances (Sasse, 1998). Additionally, CapManEx includes desludging of primary settlers, assumed to take place every 5 years for sewered systems and every 6 months for non-sewered systems (BORDA, 2016). CapManEx was allocated to the user and agency using the same allocation as used for that of CapEx (Table 4.3).

4.3.3.4 Excluded costs

Costs defined by WASHCost (2011) that are not considered in the present analysis due to insufficient information or consideration as outside the scope of the study include: (1) cost of capital; (2) expenditure on direct support; (3) expenditure on indirect support; and (4) depreciation of assets.

4.3.3.5 Allocation of costs to user and agency (summary)

Table 4.3: Allocation of CapEx, OpEx, and CapManEx costs and benefits to user and agency (utility) for financial analysis

Cost category	User: Costs (-) and benefits (+)	Agency (utility) Costs (-) and benefits (+)
CapEx	(-) Cost of household sewer connection (sewered system) (-) Cost of collection/storage (pit	(-) Cost of main conveyance infrastructure (sewered or vehicular), treatment infrastructure,

[53] The mean lifespan assumed for on-site pit latrines is the average of a range of values reported in the literature (i.e. 8 to 20 years). The effect of this wide range of latrine lifespans is considered in a sensitivity analysis for each system analyzed.

	latrine) (non-sewered system)	and disposal or reuse infrastructure
OpEx	(-) Water supply tariff (-) Pit emptying charge; sanitation service fee or surcharge (-) Fixed charge for water and sewerage connection	(-) Labor, material and energy costs (-) Water supply tariff (+) Water supply tariff (+) Pit emptying charge; sanitation service fee or surcharge (+) Fixed charge for water and sewerage connection (+) Energy, water or bio-solids co-product revenue (+) Carbon financing revenue
CapManEx	(-) Cost of collection/storage (pit latrine) replacement	(-) Cost of major repairs or replacement for conveyance, treatment and disposal/reuse infrastructure

4.3.4 Sensitivity and uncertainty analysis

A sensitivity analysis was undertaken to determine which model variables had the greatest effect on results. Each variable is individually adjusted by (+/-) 20% from baseline values. Variables are then ranked from greatest to least effect on model result and plotted on a tornado diagram (see Appendix B.2).

The variables found to have the greatest effect on model results in the sensitivity analysis are then included in an uncertainty analysis. A low and high uncertainty range is identified for each variable based on varying methods: (1) a 95% confidence interval where data are available, (2) a range of values according to site characteristics or (3) a range of values according to literature. References and explanations for baseline and uncertainty values for each variable are reported in Appendix B.3. When a 95% confidence interval was used for the uncertainty range, a Shapiro-Wilk test of normality test is undertaken (Appendix B.4). If the p-value is less than 0.05, the data are assumed to not be normally distributed; in these cases, a bootstrap confidence interval for the mean is computed in R statistical software (R Core Team, 2017).

4.3.5 Case studies

One decentralized sewered (DS) and one non-sewered (NS) sanitation system in both Zambia and India (four systems total) are evaluated to demonstrate use of the LCA model. The system type and location are selected to highlight key differences between systems and across varying environmental, institutional and social characteristics. All model input data and sources are shown in Appendix B.1.

4.3.5.1 Decentralized sewered (DS) system, Zambia

The decentralized sewered system in Zambia serves a middle-income, peri-urban community of approximately 80 households in Solwezi, Northwestern Province, Zambia. Annual temperatures range from mean lows of 5-7°C in the cold/dry season to mean highs of 26-31°C in the hot/wet season (climate-data.org). The SSC begins with private household cistern-flush toilets connected to a small-bore sewer system (Figure 4.2). Private in-house water connections are supplied *via* a piped network with water sourced from utility operated boreholes and treated surface water stored in overhead tanks. The decentralized wastewater treatment system (DEWATS) consists of five fixed-dome anaerobic digesters (ADs) ranging in volume from 22 m^3 – 36 m^3 for primary treatment; a 258 m^3 volume anaerobic baffled reactor (ABR) for secondary treatment and two 160 m^2 horizontal sub-surface flow constructed wetlands for tertiary treatment. Wastewater effluent is discharged into a nearby stream with no reuse of wastewater effluent reported. Biogas is recovered from the ADs and piped to select nearby households equipped with stoves modified for biogas use. Households without biogas connections use a mix of charcoal and electricity (at approximately a 3:1 ratio) as conventional fuel sources (Laramee et al., 2017). No bio-solids recovery is reported.

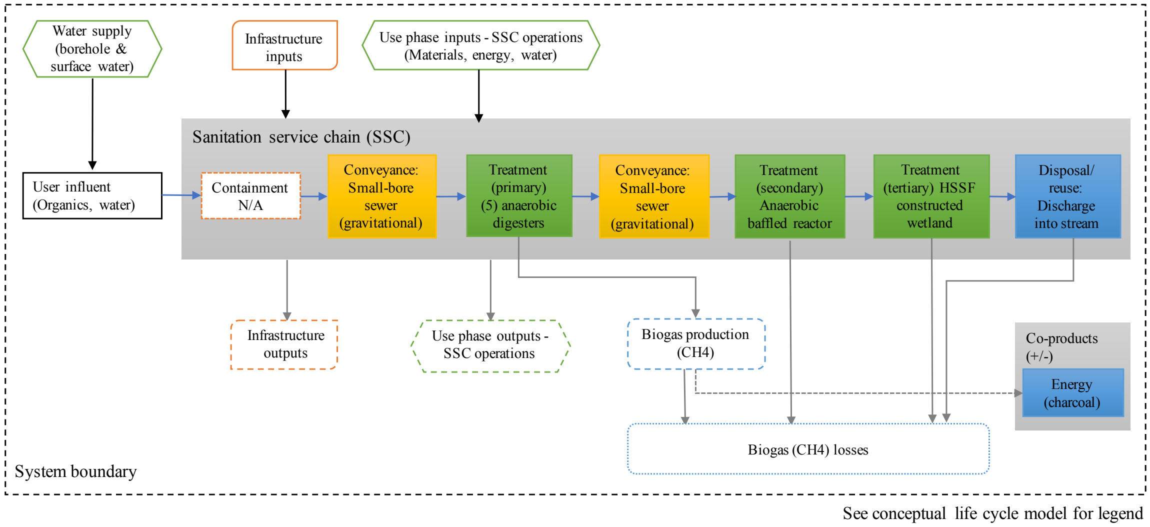

Figure 4.2: Sanitation service chain and life cycle assessment system boundary for the decentralized sewered system analyzed in Zambia (Kandundu C Compound, Solwezi, Zambia)

The ADs are constructed with cast-in-place reinforced concrete foundations and locally burnt bricks and cement plaster domes; the ABR is constructed with cast-in-place reinforced concrete with polyvinyl chloride (PVC) pipes connecting adjacent baffles; and the horizontal sub-surface flow constructed wetland consists of concrete block walls with a high-density polyurethane (HDPE) liner filled with 10-20 mm crushed stones and planted with *Canna indica*. The gravitational sewer conveyance system consists of 110 mm to 160 mm PVC pipes with concrete block manholes.

The water supply and sanitation systems are owned and operated by a commercial water utility, the Northwestern Water Supply and Sewerage Company (NWWSSC). The sanitation system has been operating since 2012. Households are billed monthly by the utility for water and sewerage services including a volumetric charge (increasing block tariff) for water supply, a sewerage surcharge based on volume of water used and a monthly fixed connection charge (see figure in Appendix B.5). At the time of data collection, households with biogas connections did not pay for biogas supply.

Data were obtained *via* primary data collection from March–August 2015; personal communication with and monitoring reports from the Bremen Overseas Research and Development Association (BORDA), a technical advisor for the sanitation project design, implementation and monitoring; personal communication with NWWSSC; and secondary literature sources. [54]

4.3.5.2 Decentralized sewered (DS) system, India

The decentralized sewered system in India serves a low-income, peri-urban community of approximately 120 households on the outskirts of Bangalore, India. Temperatures range from mean lows of 14-15°C to mean highs of 32-34°C (climate-data.org). The SSC begins with private household pour-flush toilets connected to a small-bore sewer system (Figure 4.3). Water is supplied *via* a community operated deep borewell; since 2012, supply has been supplemented with water tankers due to a subsidence of the groundwater table (Pradeep et al., 2012). Currently, the community obtains approximately equal volumes from each source (CDD, 2016). The system includes two fixed-dome 29 m^3 anaerobic digesters (ADs) for primary treatment; a 109 m^3 volume ABR for secondary treatment and a 220 m^2 horizontal sub-surface flow constructed wetland for tertiary treatment (Miller, 2011; Reynaud, 2014). Wastewater effluent is partially reused for agriculture on an experimental basis with the excess discharged into an open sewer system (Reynaud, 2015; CDD, 2016). Biogas is recovered from the ADs and piped to a community school kitchen equipped with stoves modified for biogas use (CDD, 2016). Propane was used at the school prior to biogas availability. No bio-solids reuse is reported.

[54] Primary data on site characteristics and anaerobic digester performance was collected during for the second study within this thesis (Site "A" within that study). Detailed data collection methods are described therein. Reference to data collected is denoted as Laramee et al., 2017.

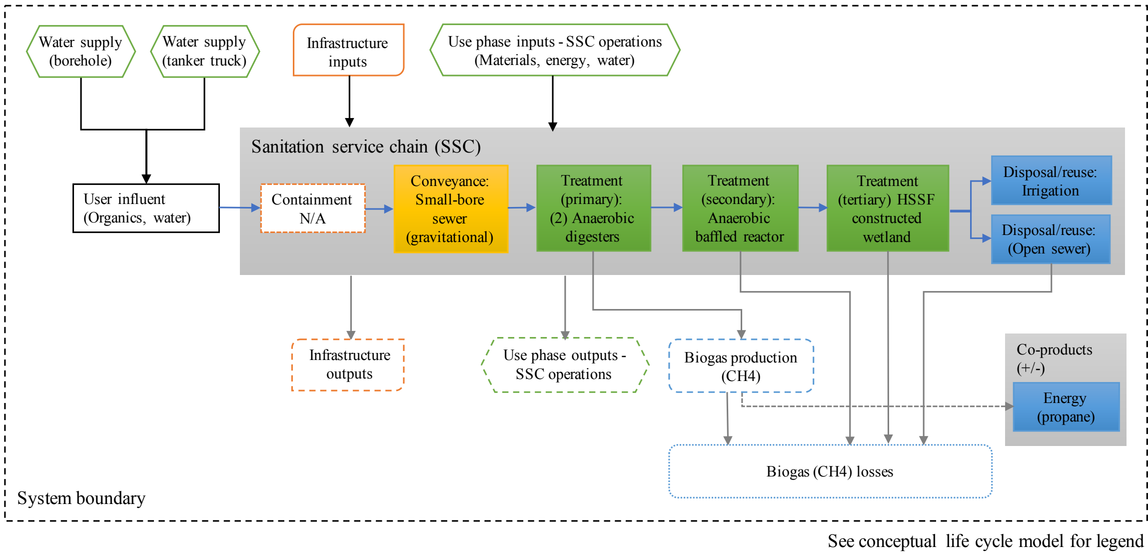

Figure 4.3: Sanitation service chain and lifecycle assessment system boundary for the decentralized sewered system analyzed in India (Beedi Workers Colony, Bangalore, India)

The ADs are constructed with cast-in-place reinforced concrete foundations and concrete block and cement plaster domes; the ABR is constructed out of concrete masonry blocks with cement plaster with PVC pipes connecting adjacent baffles; and the constructed wetland consists of concrete block walls with a high-density polyurethane (HDPE) liner filled with 10-20 mm crushed stones and planted with *Canna indica*. The gravitational sewer conveyance system consists of 110 mm diameter PVC pipes and 150 mm diameter concrete pipes with concrete block manholes.

The sanitation system, implemented in 2007, is operated in partnership between the community and the non-profit organization, the Consortium for DEWATS Dissemination (CDD), which also uses the system for research activities (CDD, 2016). The municipality performs non-recurrent maintenance activities on an as-needed basis. The community pays for the cost of electricity required to pump water from the borewell and pays private operators for water supplied *via* tanker trucks. Households do not pay for the sanitation service. Biogas is supplied to the community school at no charge.

Data were obtained *via* construction bills of quantities, monitoring data, reports and personal communication with CDD, the entity responsible for the design, implementation and monitoring of the system, and *via* literature published on the system performance by Miller (2011), Reynaud (2014) and Buckley and Reynaud (2015).

4.3.5.3 Non-sewered (NS) system, Zambia

The non-sewered system in Zambia serves the low-income, peri-urban community of Kanyama in Lusaka, Zambia. Temperatures range from mean lows of 9-11°C in the cold/dry season to mean highs of 26-31°C in the hot/wet season (climate-data.org). Households typically use simple, dry pit latrines, which are often shared between 3-4 households (Figure 4.4) (WSUP, 2016). Due to the characteristic dolomite rock formations in Kanyama, pit latrine sub-structures are typically built partially above ground with a concrete block lining. Households obtain water from public kiosks with water sourced from utility operated boreholes stored in overhead tanks (KWT, 2016). The household pit latrines are emptied manually with modified shovels and gardening tools.[55] The emptied fecal sludge is transported in 60 L barrels on manually-pushed carts to the primary treatment facility. The fecal sludge treatment plant system consists of a 58 m^3 volume AD for primary treatment and eight 35 m^2 sludge drying beds for secondary treatment (WASAZA/BORDA, 2013). Secondary conveyance *via* a vacuum tanker is required to transport the partially treated sludge approximately 7 km between primary and secondary treatment facilities as they are located at separate sites. Dried bio-solids are occasionally sold for use as a soil amendment. Biogas is recovered from the AD and is reused at the adjacent utility office for cooking. Charcoal was used exclusively prior to biogas availability.

[55] Mechanical pumping has been trialed, but due to considerable amounts of solid waste also disposed in pits has not been successful to date (WASAZA/BORDA, 2013).

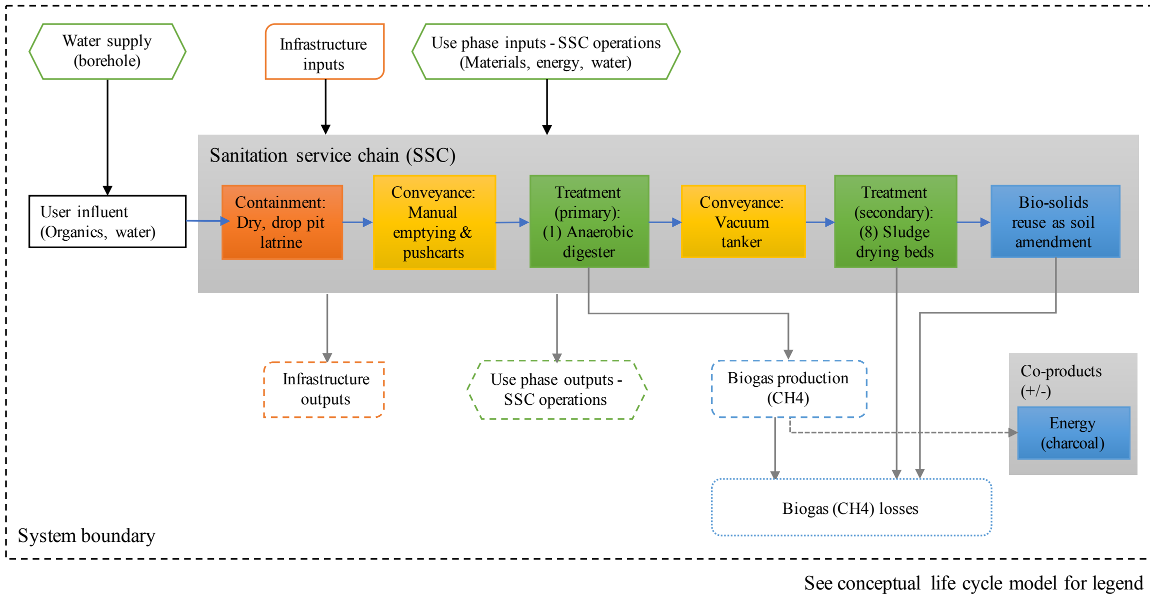

Figure 4.4: Sanitation service chain and life cycle assessment boundary for the non-sewered system analyzed in Zambia (Kanyama, Lusaka, Zambia)

The AD is constructed with a cast-in-place reinforced concrete foundation and burnt bricks and cement plaster dome. The sludge drying beds are constructed with burnt clay bricks over a sand lining, concrete block walls, and a steel and timber roof structure with transparent plastic roof sheeting.

The water supply system (kiosks, boreholes, etc.) and fecal sludge treatment plant are owned by the commercially run Lusaka Water and Sewerage Company (LWSC). The Kanyama Water Trust (KWT), managed by the LWSC, is responsible for daily water and sanitation operations. Households pay the KWT for pit emptying services on a *per* request basis and *per* volume of fecal sludge emptied. The team of pit emptiers is then paid by the KWT on a commission basis according to the volume of sludge emptied *per* month (see figure in Appendix B.5). Informal pit emptiers also continue to operate within the community; fecal sludge emptied from pits by informal emptiers is generally disposed of underground in a hole dug next to the pit structure, which was the common practice prior to implementation of the fecal sludge treatment plant (KWT, 2016).

Data used in the LCA model were obtained *via* construction bills of quantities, monitoring data, reports and personal communication with the Water and Sanitation Association of Zambia (WASAZA) and BORDA, technical advisors for the sanitation project design; Water and Sanitation for the Urban Poor (WSUP), the overall project coordinator; and the KWT, the local operator of the system. Secondary literature sources were also used for some model parameters.

4.3.5.4 Non-sewered (DS) System, India

The non-sewered system in India serves the peri-urban community of Devanahalli, located on the north-east outskirts of Bangalore, India. Temperatures in Bangalore range from mean lows of 14-15°C to mean highs of 32-34°C (climate-data.org). No centralized sewer or wastewater treatment plant exists in Devanahalli (CDD, 2016). Households typically use pour-flush toilets connected to single pits, generally constructed out of stone masonry or reinforced concrete rings (Figure 4.5) (CDD, 2016). Water is sourced from deep borewells and supplied to households *via* a piped network (2 – 3 days *per* week) operated by the municipality or *via* privately operated tanker trucks (4 – 5 days *per* week) when piped water is not available (CDD, 2016).

Pit emptying and conveyance of fecal sludge is *via* motorized vacuum tankers operated by the municipality and private operators. Prior to the implementation of the treatment plant, vacuum tanker operators generally discharged fecal sludge directly from pits onto nearby farm lands. The first step of the treatment plant is a feeding tank used for separation of liquids and solids. After separation, the solid fraction is treated in two 6 m^3 biogas digesters operated in parallel, two 12 m^3 stabilization tanks and ten 12 m^2 sludge drying beds. Recovered biogas is used by the office and caretaker at the treatment plant. Dried bio-solids are then co-composted with municipal solid waste and occasionally sold as a soil amendment. The liquid stream is treated *via* one 12 m^3 anaerobic baffled reactor and a 10 m^2 horizontal sub-surface flow constructed wetland, with effluent disposed of in a percolation pit (all details on system from CDD, 2016).

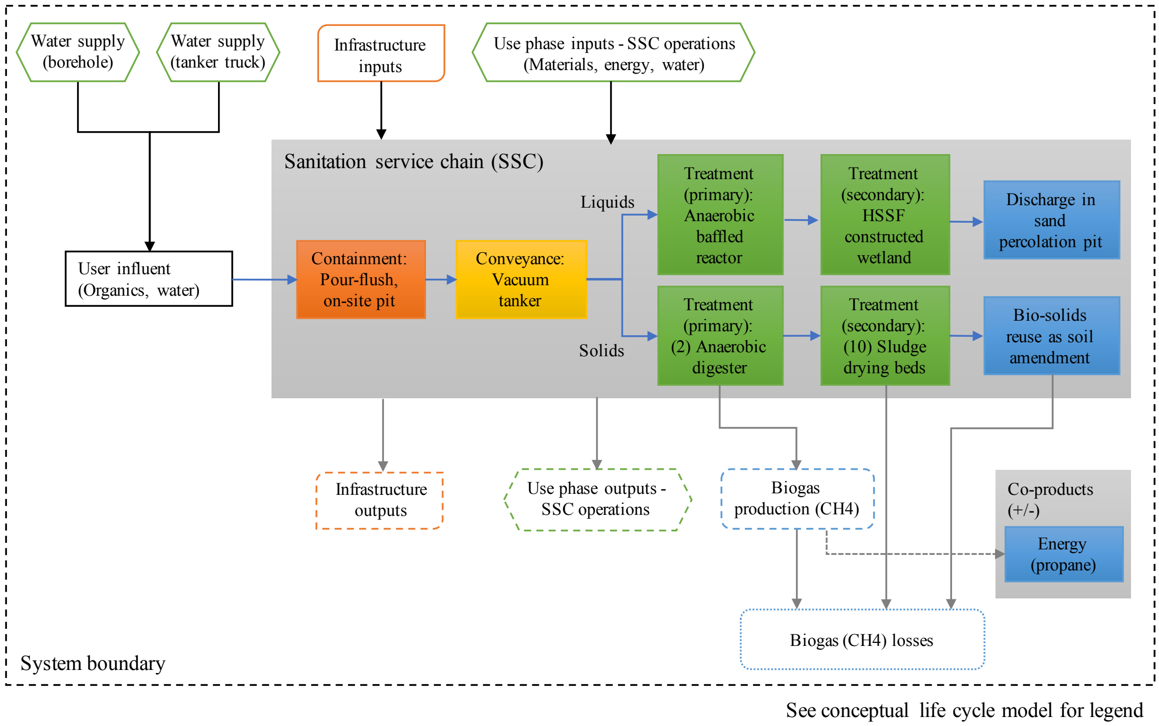

Figure 4.5: Sanitation service chain and lifecycle assessment boundary for the non-sewered system analyzed in India (Devanahalli, Bangalore, India)

The treatment modules are constructed primarily from pre-fabricated fiberglass units. The sludge drying beds are constructed with cement plaster concrete block walls, a burnt brick base overlaying a stone and sand filter, with a steel roof structure and transparent plastic roofing (CDD, 2016).

The sanitation system, which was commissioned in late 2015, was operated in partnership by CDD and the local municipality for approximately the first year of operation. Thereafter, the municipality has gradually assumed operational responsibilities. Households pay either the municipality or private operators for pit emptying services on a *per* request basis. Private operators discharge fecal sludge at the treatment plant at no charge. Households pay a fixed monthly charge to the utility for municipal piped water and pay private tanker truck operators *per* volume of water.

Data were obtained *via* construction bills of quantities, monitoring data and personal communication with CDD, the organization responsible for the design, implementation and management of the system in the start-up and first year of operations. Data were also obtained *via* secondary literature sources.

4.4 Findings

4.4.1 Site characteristics and water use

Study site characteristics and key LCA model inputs are reported in Table 4.4 (see Appendix B.1 for complete reporting of model inputs for each case study). Notably, the non-sewered systems are estimated to serve a significantly greater population compared to the decentralized sewered systems. The range of uncertainty for the number of people served by the non-sewered systems – calculated using mean organic matter produced *per capita per* day, mean total solids content and mass of fecal sludge discharged into the treatment plant *per* day *via* methods outlined previously – is also considerably greater.

Water use for sanitation is significantly higher *per capita* for sewered systems *versus* non-sewered systems. Differences between systems are particularly pronounced in Zambia with an annual *per-capita* water use of $27m^3$/*capita*/year for the sewered system and $<1m^3$/*capita*/year for the non-sewered system. For the India case studies, water use is $10m^3$/*capita*/year and $3m^3$/*capita*/year for sewered and non-sewered sanitation systems, respectively. If water losses during distribution are considered, the total amount of water supplied from point of extraction may be 1.5 to 2 times higher than these figures.[56]

Water use is also considerably higher (2.5 times higher) for the sewered system in Zambia *versus* India. The higher *per-capita* wastewater volume results in a lower COD concentration (832 mg/L *versus* 1,871 mg/L for the sewered systems in Zambia and India, respectively), which may explain the lower primary treatment efficiency for the sewered Zambia system (Laramee et al., 2017). Treated wastewater from the system in

[56] Non-revenue water is reported to be 49% on average for water and sewerage utilities across Zambia and 36% for the Bangalore Water Supply and Sewerage Board (NWASCO, 2017; IBNET, 2009).

India is partially reused for irrigation; however, reuse is experimental and not considered as a replacement for conventional water. No water reuse is reported for the sewered system in Zambia.

In contrast, water use is lower for the non-sewered system in Zambia *versus* India. Non-sewered system users in Zambia typically use dry, drop pit latrines requiring no water. In contrast, users of the non-sewered system in India typically use pour-flush toilets connected to on-site pits. The use of flush water in India also results in a considerably lower total solids (TS) content of fecal sludge as compared to the Zambia system. With a higher TS content, the number of people served by the system in Zambia is estimated to be approximately twice that served by the India system despite receiving only half of the total wet mass of fecal sludge *per* day.

Treatment infrastructure volume and land area are considerably greater for the sewered *versus* non-sewered systems with 25 – 200 times greater *per-capita* reactor volumes and 8 – 16 times *per-capita* land area required. *Per-capita* wastewater generation influences the sizing of treatment infrastructure for both system types.[57] With a greater wastewater generation rate, the sewered system in Zambia has a 2.5 times greater reactor volume and 2.2 times the land area compared to the sewered system in India. Similarly, with flush water use, the non-sewered system in India has a 3 times greater reactor volume compared to the non-sewered system in Zambia; land area is, however, similar between the two non-sewered systems.

An MCF of 0.7 has been assigned for the pit containment system in India due to the use of flush water, whereas an MCF of 0.5 has been assigned for the pit latrines in Zambia for communal pits in dry conditions (IPCC, 2006). Thus, the pit conditions for the India system are assumed to be more anaerobic and convert greater proportions of COD to CH_4 during degradation. The percentage of organic loads removed in the pit containment systems in Zambia and India are assumed to be equal based on methods described

[57] The key design parameter for the decentralized sewered treatment systems is the up-flow velocity of wastewater through the chambers of the anaerobic baffled reactor (Gutterer et al., 2009). Thus, with greater volumes of wastewater, greater reactor volumes are required.

previously. Although the initial degradation kinetics may differ due to variations in aerobic *versus* anaerobic conditions, several years are reported between emptying events in both systems, thus it is assumed that the long-term extent of degradation will be similar.

To compare systems on a consistent basis, the extent of biogas recovery from primary treatment modules is assumed to be equivalent across systems in the baseline analysis. An 82% biogas recovery is assumed based on the mean energy recovery found across fifteen systems in three communities in Zambia (Laramee et al., 2017). Notably, the alternative fuel replaced by biogas in Zambia is primarily charcoal, whereas propane gas is the primary conventional fuel used for cooking in India.

Table 4.4: Key site characteristics and life cycle assessment (LCA) model inputs for each case study system analyzed

Parameter	**Dec. Sewered, Zambia**	**Dec. Sewered, India**	**Non-sewered, Zambia**	**Non-sewered, India**
	Mean 95% Conf. Interval [2.5, 97.5] or (uncertainty range)	Mean 95% Conf. Interval [2.5, 97.5] or (uncertainty range)	Mean 95% Conf. Interval [2.5, 97.5] or (uncertainty range)	Mean 95% Conf. Interval [2.5, 97.5] or (uncertainty range)
Number of people served	394 [a] (374 - 414) [b]	575 [e] (546 - 604) [b]	12400 [i] (6400 - 23700) [i]	6900 [i] (4000 - 12300) [i]
Number of people *per* latrine	5.0 [a] [4.5, 5.4] [a]	4.8 [f] (4.6 - 5.0) [b]	5.0 [a, j] [4.5, 5.4] [a, j]	4.4 [h] (4.2 - 4.6) [h, b]
Wastewater generation *per capita* (LPCD)	74 [a] [69, 80] [a]	29 [e] [27, 30] [e]	0 [k] -	7 [h, q] (4 - 10) [h, q]
COD generation *per capita* (gPCD)	62 [a] [56, 67] [a]	54 [g] [52, 55] [g]	62 [a, l] [56, 67] [a, l]	54 [g, l] [52, 55] [g, l]
Total solids content (%)	- -	- -	16.5% [m] [11%, 22%] [m]	4.2% [h] [3.2% - 5.3%] [h]
Fecal sludge input *per* day (kg wet weight)	- -	- -	912 [m] [799, 1025] [m]	1978 [h] [1232 - 2724] [h]
Type of toilet and containment	Cistern flush, private [a]	Pour flush, private [h]	Dry, drop pit latrine, shared [n]	Pour flush with pit, private [h]
COD removal from pit latrine	- -	- -	88% [o] (85% - 91%) [o]	88% [o] (85% - 91%) [o]
Treatment reactor volume *per capita* (m^3/*capita*)	2.00 [c]	0.76 [g]	0.01 [c]	0.03 [h]
Primary treatment (% COD removal)	61% [a] [58%, 63%] [a]	73% [g] [68%, 77%] [g]	22% [m] [19%, 26%] [m]	5% [h] [3%, 7%] [h]
Biogas energy recovery (%)	82% [a] [76%, 88%] [a]	82% [a, p] [76%, 88%] [a, p]	82% [a, p] [76%, 88%] [a, p]	82% [a, p] [76%, 88%] [a, p]
Conventional fuel used	Charcoal (78%), Electricity (22%)[a]	Propane gas [h]	Charcoal [n]	Propane gas [h]
MCF (pit containment)	- -	- -	0.5 [d] (0.25 - 0.75) [d]	0.7 [d] (0.35 - 1.0) [d]
MCF (primary treatment)	0.9 [d] (0.8 - 1.0) [d]	0.9 [d] (0.8 - 1.0) [d]	0.9 [d] (0.8 - 1.0) [d]	0.9 [d] (0.8 - 1.0) [d]
Treatment infrastructure land area (m^2/*capita*) [r]	2.0 [c] (1.9 - 2.1) [c]	0.90 [h] (0.85 - 0.94) [h]	0.12 [n] (0.06 - 0.23) [n]	0.11 [h] (0.06 - 0.18) [h]

[a] Laramee et al. (2017)
[b] Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006)
[c] BORDA (2016)
[d] Table 6.3 and Table 6.7 (IPCC, 2006)
[e] Reynaud and Buckley (2015)
[f] Reynaud (2014)
[g] Computed with data from Reynaud and Buckley (2015) and Miller (2011)
[h] CDD (2016)
[i] Calculated based on mean and range of kg fecal sludge inputs to the treatment plant *per* day, total solids content (%) and dry weight of organic solids produced *per capita per* day (see methods)
[j] Measured mean number of people *per* latrine: 17.7 (SD=8.6) (WSUP, 2016). However, mean number

of people *per* household from DS, Zambia used in baseline analysis for standard comparison across systems and to meet SDG criteria for improved sanitation (not shared).
[k] Assume no water used for sanitation (household toilets typically dry pit latrines) (KWT, 2016).
[l] *Per capita* COD assumed to be equivalent to COD *per capita* measured for sewered system (no data available specific to site)
[m] WASAZA/BORDA (2013)
[n] KWT (2016)
[o] Computed as a function of COD content at varying pit depths with assumed 1m depth removed during pit emptying (see methods): Figure 1 - Nwaneri et al. (2008).
[p] Biogas energy recovery (%) is assumed to be equal across projects for a standard basis of comparison. Value is based on % recovery found in Laramee et al. (2017).
[q] Mara (1981)
[r] Land area for sewered systems does not include treated effluent infiltration area as after treatment effluent is discharged into a stream (Zambia) and into an open sewer (India).

4.4.2 Energy use

The majority of *per-capita* energy use (77%) for the sewered system in Zambia is incurred as infrastructure embodied energy (Figure 4.6 and Figure 4.8). Energy required for water supply in the use phase is the second greatest contributor of lifecycle energy use (13%). Embodied energy use for infrastructure of the sewered system in India is comparatively less than for the system in Zambia due to the smaller *per-capita* reactor volume: 95 MJ/*capita*/year *versus* 224 MJ/*capita*/year, amounting to 40% of lifecycle energy use. Among the SSC functional groups, treatment infrastructure contributes the greatest impact for both sewered systems, accounting for 64% and 67% of embodied energy use for the Zambia and India system, respectively. Despite the lower wastewater generation by users of the sewered system in India, energy use associated with water supply accounts for 56% of overall lifecycle use and is over three times higher in India (133 MJ/*capita*/year *versus* 39 MJ/*capita*/year). Minimal energy is required for treatment operations during the use phase for both sewered systems as treatment is passive and primarily biological, thus requiring no regular energy inputs.

For non-sewered systems, *per-capita* infrastructure embodied energy is considerably lower relative to the sewered systems in both Zambia and India: 25 MJ/*capita*/year and 28 MJ/*capita*/year, amounting to 58% and 37% of lifecycle energy use, respectively. Nearly all infrastructure embodied energy is due to on-site containment (pit latrine substructures), accounting for 84% of embodied energy for the Zambia system and 69%

for the India system. In contrast to the sewered systems, relatively little energy is used for treatment infrastructure: 40 times and 10 times lower for the Zambia and India non-sewered systems, respectively. This can be attributed to the different characteristics of waste discharged into the sewered *versus* non-sewered systems, thus requiring different treatment: fecal waste undergoes considerable degradation in pits and therefore enters the treatment plant in a partially degraded, concentrated form as compared to the sewered systems, which treat dilute, non-degraded waste. Similar to the sewered systems, energy required for water supply is greater for the non-sewered system in India *versus* Zambia, accounting for 31% of lifecycle energy use due to the higher energy intensity for water supply and higher *per-capita* water use. Energy use for emptying and conveyance of fecal sludge for both non-sewered systems accounts for 18% of lifecycle energy use for the system in Zambia and 24% of lifecycle energy use for the system in India.

A greater potential to reduce energy use *via* biogas energy recovery is observed for the sewered systems *versus* non-sewered systems. Specifically, the sewered system in Zambia is found to have the greatest energy offsets due to the replacement of primarily charcoal, an inefficient fuel source. Less potential to offset energy use is observed for the sewered system in India due to the conventional use of comparably higher efficiency propane. Therefore, although the sewered system in Zambia has the highest energy use requirements for infrastructure and operations, given the potential energy offsets of charcoal with recovered biogas, this system has the lowest lifecycle energy use when more than 70% of biogas is recovered and has a net negative energy impact at biogas energy recovery levels over 76% (Figure 4.10).

In contrast, minimal lifecycle energy reduction benefits are observed by increasing biogas energy recovery from the primary treatment stage for the non-sewered systems. Considerable degradation and release of methane occurs during containment and storage in pits, where no mechanism to capture methane exists. Energy recovery offsets are found to be slightly higher for the non-sewered system in Zambia *versus* India due to (1) the replacement of charcoal, an inefficient energy source, and (2) a greater primary treatment reactor volume that yields greater COD reduction and biogas production as compared to the treatment system in India. However, only minimal energy recovery potential exists compared to the sewered system in Zambia – 17 MJ/*capita*/year *versus* 304 MJ/*capita*/year at the baseline assumption of 82% biogas recovery. Energy recovery *via* use of bio-solids is not considered in the current lifecycle model and may be an important source of energy benefits, particularly for non-sewered systems.

Although at baseline energy recovery levels, the sewered system in Zambia is likely to have the lowest lifecycle energy use, it also has the highest levels of uncertainty (Figure 4.6 and Figure 4.8). The greatest contributors to uncertainty for the sewered Zambia system are COD generation *per capita*, primary treatment efficiency, percent of biogas energy recovery and fuel efficiencies of biogas and charcoal (Appendix B.2 and B.3). As the extent of energy offsets *via* biogas recovery is dependent on these variables, the greatest uncertainty for the sewered Zambia system is associated with the disposal/reuse functional group (Figure 4.6). Contributors to uncertainty are similar for the sewered system in India, except for fuel efficiencies as propane is assumed to have an equivalent efficiency to biogas. Variables associated with water supply, such as the *per-capita* wastewater generation and the distance traveled by water supply trucks are also found to be considerable sources of uncertainty for the India sewered system.

A comparably smaller uncertainty range is found for the non-sewered systems, particularly in Zambia. The number of people *per* latrine and the estimated life-span of on-site pit infrastructure are found to be among the highest contributors of uncertainty for the non-sewered systems as both variables affect the *per-capita* embodied energy for pit infrastructure. The percent of organic load removed in the pit infrastructure is the greatest source of uncertainty for the non-sewered system in Zambia as lower organic removal in the containment stage leads to greater potential energy recovery during primary treatment and thus greater potential to offset conventional energy.

4.4.3 Global warming potential (GWP)

In contrast to energy use, nearly all lifecycle GWP impacts (kgCO_2e emissions *per capita*) occur during the use phase for both sewered and non-sewered systems (Figure 4.9). The largest contributor to GWP impacts for all systems is CH_4 emissions resulting from anaerobic degradation of waste, accounting for 67% and 71% of lifecycle GWP for the sewered systems, and 87% and 77% for the non-sewered systems in Zambia and India, respectively.

Nearly all CH_4 emissions occur in the treatment stage of the sewered systems (>99%) (Figure 4.7). At the baseline assumption for biogas energy recovery, lifecycle emissions are reduced by 35% and 48% *via* CH_4 recovery and by 51% and 7% *via* conventional energy replacement for sewered systems in Zambia and India, respectively. Greater emissions reduction in Zambia are due to the replacement of conventionally used charcoal, which produces substantial GHG emissions during production and use. In contrast, nearly all emissions occur in the containment stage of the non-sewered systems (>92%) (Figure 4.7). As no mechanism is available for CH_4 capture during containment in on-site pit latrines, minimal potential exists to reduce emissions *via* CH_4 recovery or replacement of conventional fuels. Combined recovery of CH_4 from the primary treatment stage, and offset of conventional fuels, reduces lifecycle emissions by 9% for the non-sewered system in Zambia and less than 1% in India.

Thus, similar to energy use, at low percentages of biogas recovery, the sewered systems are observed to have the highest GWP impact, while at high rates of energy recovery, the sewered systems have the lowest GWP impact as these systems have the greatest potential to capture and offset emissions (Figure 4.11). The decentralized system in Zambia achieves net negative lifecycle emissions above approximately 95% energy recovery. In contrast, minimal GWP benefit is achieved by increasing primary treatment energy recovery for the non-sewered systems.

Emissions resulting from conveyance of waste, either *via* waterborne sewerage or vehicular means, are minimal for all systems. Water used for sanitation accounts for 12% and 14% of lifecycle emissions for the sewered systems, while vehicular use accounts for <1% and 6% of emissions for the non-sewered systems in Zambia and India, respectively.

Similar to energy use, the sewered system in Zambia is likely to have the lowest GWP at the baseline biogas recovery of 82%, but also has the highest range of uncertainty (Figure 4.7). The percent of biogas energy recovery and primary treatment efficiency are the largest contributors of uncertainty for both the Zambia and India sewered systems as these variables affect the extent of CH_4 emissions, the largest contributor to lifecycle GWP impact. Fuel efficiencies for gas and charcoal are also an important source of uncertainty for the Zambia sewered system (see sensitivity and uncertainty analysis in Appendix B.2 and B.3).

The greatest source of uncertainty for non-sewered systems occurs at the containment stage, primarily due to the +/- 50% uncertainty range recommended by IPCC (2006) for the assumed latrine MCF. Ranges for COD generation *per capita* and organic load removed during containment are also important sources of uncertainty for both non-sewered systems (see sensitivity and uncertainty analysis in Appendix B.2 and B.3).

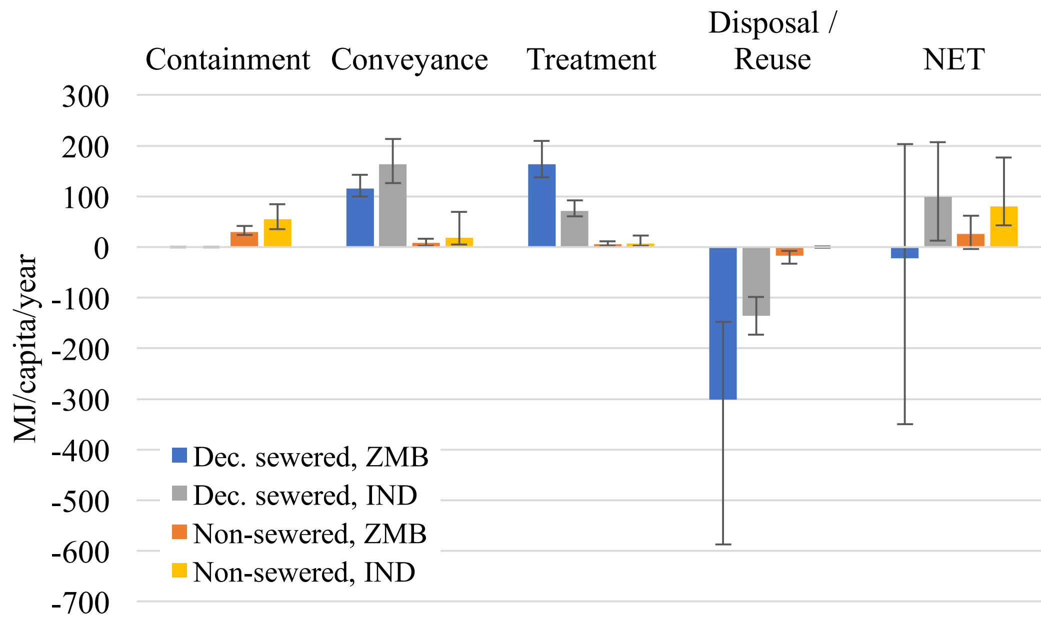

Figure 4.6: *Per-capita* energy use by functional group for infrastructure and use phase at baseline assumption of 82% biogas recovery for each case study system analyzed (MJ/*capita*/year)

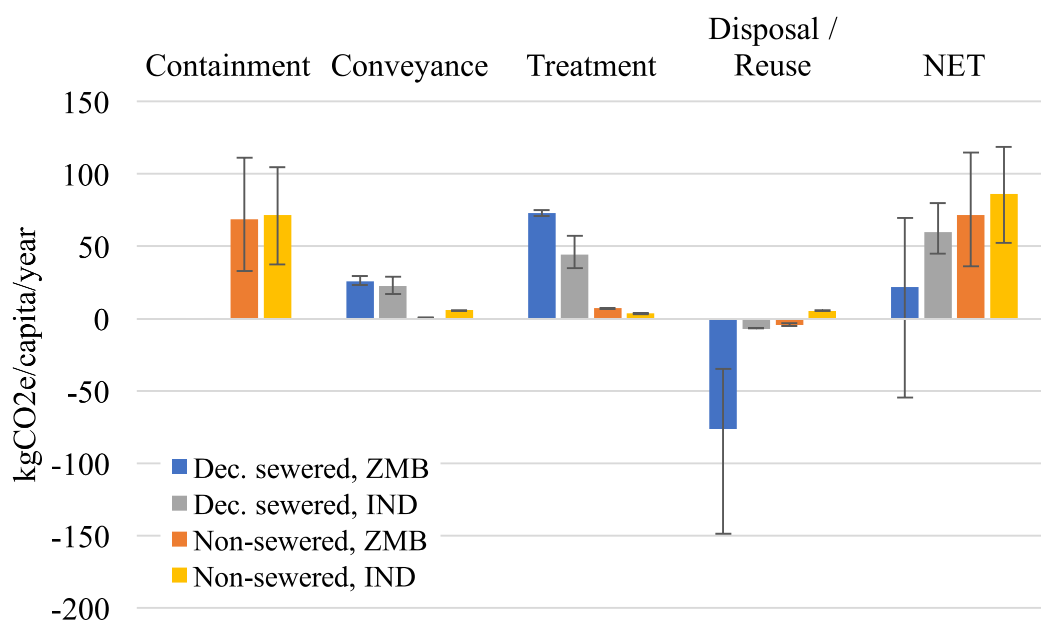

Figure 4.7: *Per-capita* global warming potential (GWP) by functional group for infrastructure and use phase at baseline assumption of 82% biogas recovery for each case study system analyzed (kgCO_2e/*capita*/year)

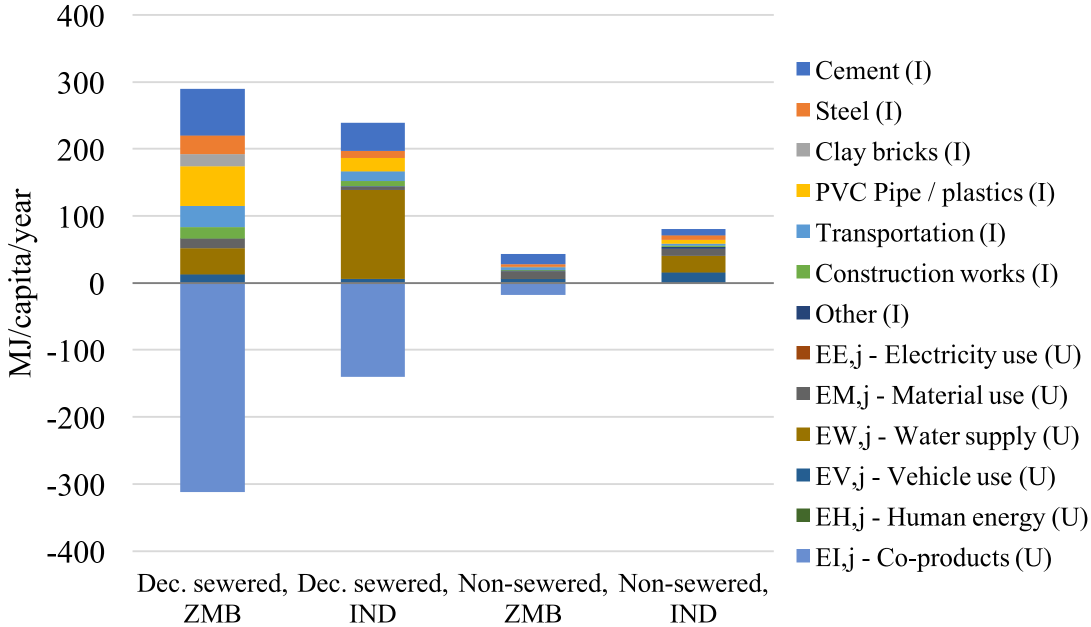

Figure 4.8: *Per-capita* energy use according to the source of impact for infrastructure (I) and use (U) phase at baseline assumption of 82% biogas recovery for each case study system analyzed (MJ/*capita*/year)

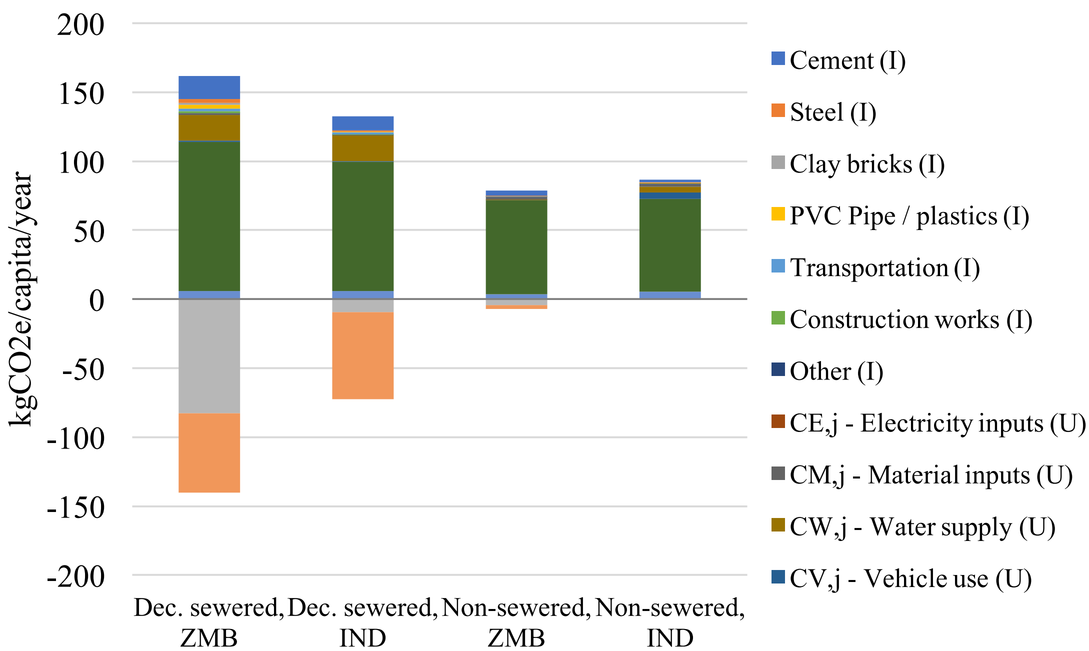

Figure 4.9: *Per-capita* global warming potential (GWP) according to the source of impact for infrastructure (I) and use (U) phase at baseline assumption of 82% biogas recovery for each case study system analyzed (kgCO_2e/*capita*/year)

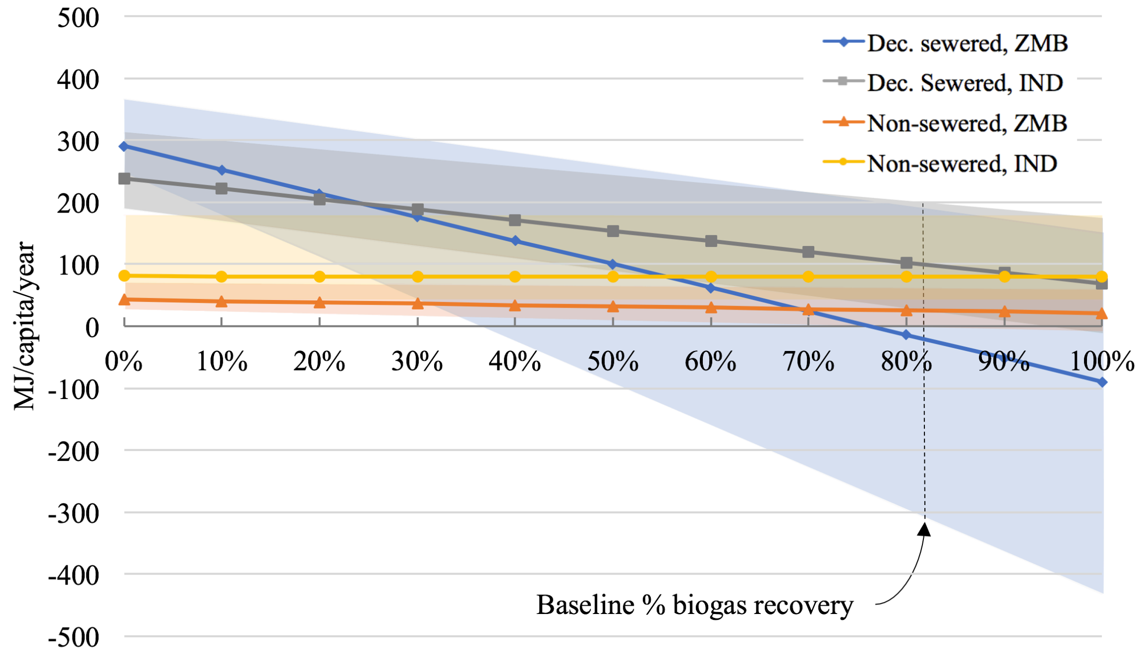

Figure 4.10: *Per-capita* energy use according to the percentage of biogas energy recovered: MJ/*capita*/year from 0% to 100% biogas energy recovery

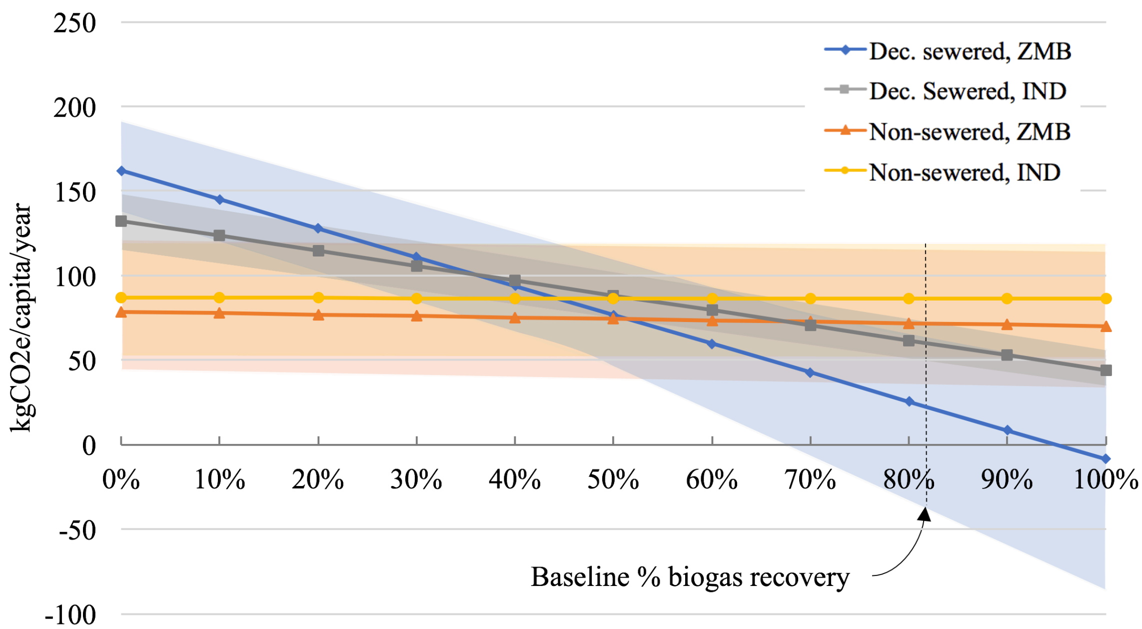

Figure 4.11: *Per-capita* global warming potential (GWP) according to the percentage of biogas energy recovered: MJ/*capita*/year from 0% to 100% biogas energy recovery

4.4.4 Economic costs and benefits

Net *per-capita* equivalent annual costs are higher for the sewered systems *versus* non-sewered systems for both the user and agency over the assumed 20-year infrastructure lifespan (Figure 4.7). Mean cost differences between systems are particularly striking in Zambia with *per-capita* user and agency equivalent annual cost 5 and 17 times higher, respectively, for the sewered *versus* non-sewered system. Mean annual costs are also higher for the sewered *versus* non-sewered system in India, although uncertainty ranges overlap.

Costs between countries vary. Mean user and agency costs for the sewered system are approximately 3 times higher *per-capita* in Zambia *versus* India. In contrast, slightly higher mean costs are found for the non-sewered system in India, with 1.2 and 1.9 times higher *per-capita* EACs for user and agency, respectively.

The greatest sources of uncertainty for users of sewered systems in both Zambia and India are the *per-capita* volume of water used for sanitation and the water supply tariff (see Appendix B.2 and B.3). For the sewered system agencies, the estimated CapEx is a critical source of uncertainty in both countries. The number of people sharing a latrine is the most important source of uncertainty for non-sewered system users in both countries. In Zambia, the lifespan and CapEx of on-site sanitation infrastructure are also important sources of uncertainty, while in India the water supply tariff is an important source of uncertainty for non-sewered system users. The greatest sources of uncertainty for the agencies of non-sewered systems are operational costs and variables affecting the calculation of number of people served.

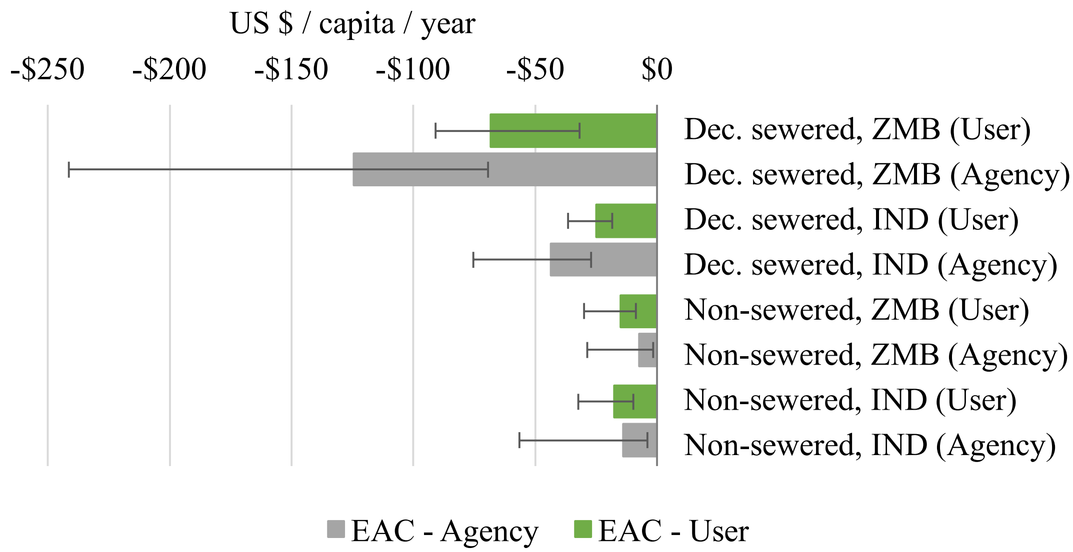

Figure 4.12: User and agency net equivalent annual cost (EAC) for each case study system with costs and benefits shown as negative and positive figures, respectively (US $, 2015)

CapEx *per capita* is also higher for the sewered *versus* non-sewered systems, particularly for the agency: sewered system CapEx is 60 times higher in Zambia and 5 times higher in India (Figure 4.13). In contrast, *per-capita* CapEx incurred by the user is greater for the non-sewered systems as users are assumed to pay for on-site containment infrastructure (e.g. pit latrine sub-structure). In other words, sewered system users pay a considerably lower proportion of upfront CapEx – 4% and 3% – *versus* the non-sewered system users – 79% and 46% of CapEx in Zambia and India, respectively.

In contrast to CapEx, *per-capita* operating costs are higher for users of the sewered *versus* non-sewered systems. Specifically, operating costs are considerably higher for sewered system users in Zambia, with combined OpEx and CapManEx EAC *per capita* 9 times higher than for non-sewered system users. Operating EACs for the agency, considering combined OpEx costs and benefits and annualized CapManEx, are also slightly higher for sewered *versus* non-sewered systems.

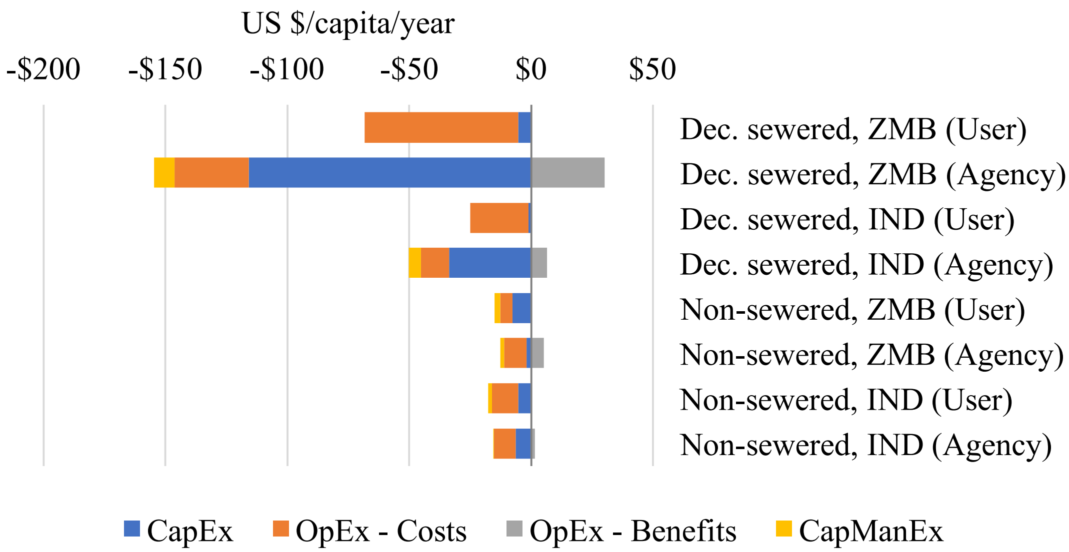

Figure 4.13: User and agency capital, operational and capital maintenance expenditures with costs and benefits shown as negative and positive figures, respectively (US $, 2015)

The greatest cost for users of sewered systems in both Zambia and India was water supply during the use phase, which accounted for 56% and 88% of overall *per-capita* EAC, respectively (Figure 4.14). Water supply was also the highest cost for non-sewered system users in India, where pour-flush toilets connected to on-site pits are typically used, accounting for 50% of overall *per-capita* EAC. CapEx was the highest cost for users of the non-sewered system in Zambia (51% of overall *per-capita* EAC).

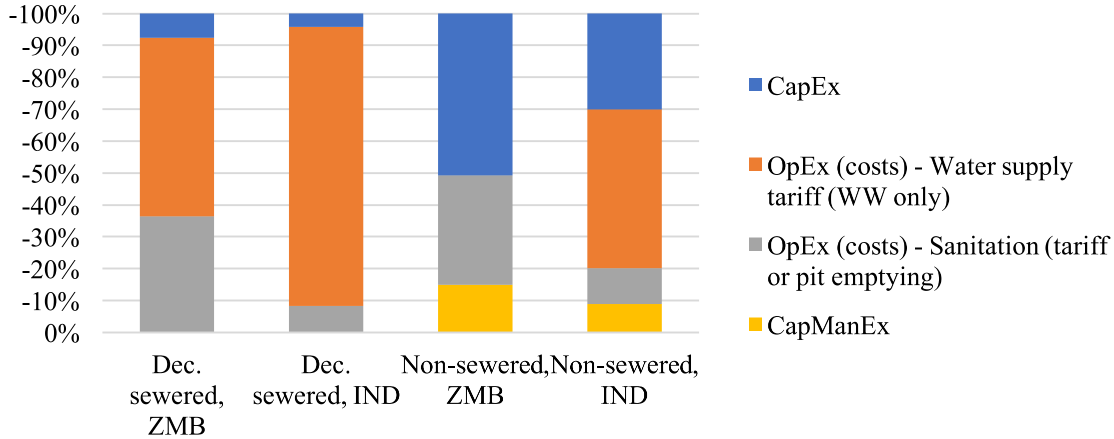

Figure 4.14: Percentage of user equivalent annual cost (EAC) for capital (CapEx), operational (OpEx) and capital maintenance (CapManEx) expenditures

For sewered system agencies in both countries, CapEx is the highest cost, accounting for 75% and 67% of overall *per-capita* EAC in Zambia and India, respectively (Figure 4.15). In contrast, labor is the greatest cost for non-sewered system agencies in Zambia (56% of overall *per-capita* EAC), while for the non-sewered system agency in India, costs for labor and CapEx are similar: 40% and 41%, respectively.

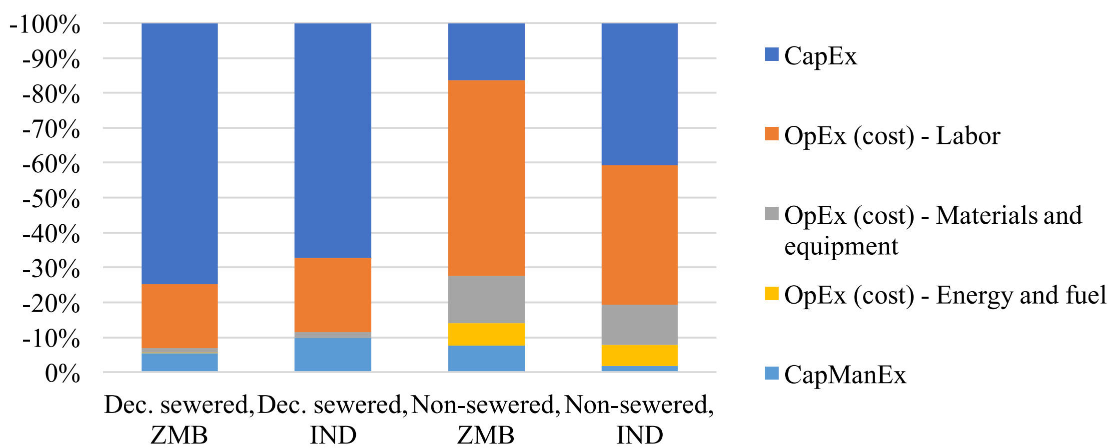

Figure 4.15: Percentage of agency equivalent annual cost (EAC) for capital (CapEx), operational (OpEx) and capital maintenance (CapManEx) expenditures

Agency benefits are highest for the sewered system in Zambia, with 82% of benefits accruing from a sanitation tariff (Figure 4.16). Nearly all benefits are attained *via* pit emptying fees for the non-sewered systems, accounting for 95% and 98% of EAC benefits in Zambia and India, respectively. The theoretical benefits of biogas recovery were minimal in comparison to total capital and operating costs, estimated at 4% of annual agency costs for the sewered system in Zambia, 9% of agency costs for the sewered system in India, 3% of agency costs for the non-sewered system in Zambia and <1% of agency costs for the non-sewered system in India.[58]

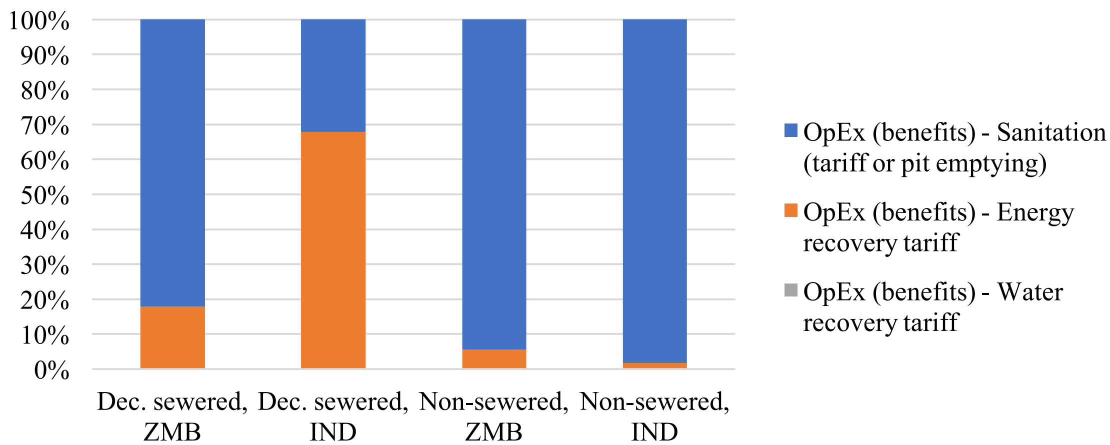

Figure 4.16: Percentage of agency equivalent annual cost (EAC) benefits for operational revenue streams: sanitation tariff, energy recovery tariff and water recovery tariff

4.4.4.1 Carbon pricing

Potential revenue generated from carbon offsets is not considered in the baseline financial analysis as none of the case studies accessed this source of funding at the time of the study. The theoretical revenue which could be accessed *via* carbon offsets is explored under three carbon pricing scenarios at the assumed baseline of 82% biogas recovery from the primary treatment unit. At the low carbon price considered, the agency equivalent annual cost is reduced by less than 1% for all systems (Figure 4.17). At the mid and high level pricing considered, the agency equivalent annual costs could potentially be reduced by 5 to 18% for the Zambia sewered system and by 8% to 29% for the India sewered system, respectively. Minimal potential exists for financial benefits *via*

[58] Financial benefit *via* biogas recovery is the theoretical savings accrued by offsetting conventional fuel sources as explained in the methods Section 4.3.3.

carbon offset from biogas recovery for the non-sewered systems as most biogas is lost during the containment stage in pit latrines.

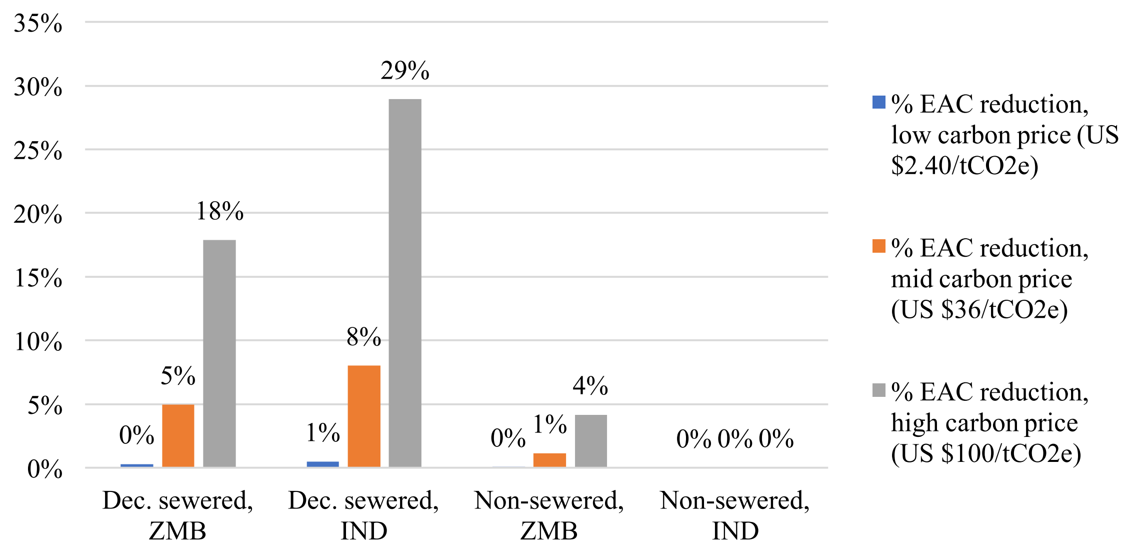

Figure 4.17: Percentage reduction of equivalent annual cost (EAC) *via* carbon pricing at varying levels of carbon pricing (US $ *per* tonne of CO_2e

4.5 Discussion

The proposed sanitation target of the Sustainable Development Goals (SDGs) is to achieve universal access to safely managed sanitation services by 2030 (UN, 2015). Major sanitation investments will be necessary for growing unserved populations, particularly in urban sub-Saharan Africa (SSA) and South Asia (SA) where 90% of urban population growth is expected to take place. In line with the proposed SDGs, which expand the sanitation focus beyond simply the toilet facility, a greater proportion of investment will be directed to the entire sanitation service chain. Evaluating resource use, GHG emission impacts and financial costs and benefits of the increasing variety of sanitation approaches is necessary to inform such sanitation investments. Within this study, a lifecycle assessment (LCA) and lifecycle cost (LCC) model is outlined to assess alternative sewered and non-sewered sanitation options. Secondly, use of this model is demonstrated to assess two decentralized sewered and two non-sewered sanitation systems operating in Zambia and India.

Through analysis of these case studies, key drivers of impacts both within and between sewered and non-sewered approaches are identified. In the context of the case study systems analyzed, findings from this study suggest (1) the significantly higher *per-capita* water use by sewered systems users, required for conveyance of waste through sewer networks, is a key driver of lifecycle energy use for sewered systems, while motorized conveyance of waste does not contribute appreciably to lifecycle energy use or GWP impacts for non-sewered systems; (2) infrastructure embodied energy is a key driver of lifecycle energy use for both sewered and non-sewered systems; (3) direct methane emissions during the use phase drives the magnitude of GWP impacts for both system types; (4) the percent of biogas recovery and baseline fuel use strongly influence lifecycle energy and GWP for sewered systems, whereas these factors had little influence for non-sewered systems; (5) lifecycle financial costs are driven by infrastructure capital costs for both system types, as well as water supply for sewered systems and labor costs for non-sewered systems; (6) financial costs are significantly higher for sewered system users and agencies; however, non-sewered system users may incur higher upfront capital costs and a greater proportion of lifecycle costs compared to sewered system users; and (7) the share of capital and operational costs is inversely distributed between users and agencies for sewered *versus* non-sewered systems suggesting different financial models are needed for different sanitation approaches. These key characteristics are discussed in more detail in the following sections.

4.5.1 Water supply and conveyance

Water use differs significantly between sanitation approaches and according to water accessibility and cultural norms. Within this study, significantly higher annual *per-capita* water use is found for sewered systems, which rely on water for conveyance, *versus* non-sewered systems. Water use is also significantly higher for the sewered system in Zambia *versus* India, likely due to the relative accessibility of water. Piped, in-house water is supplied to users of the sewered system in Zambia, whereas due to a low groundwater table and unreliable electricity supply, users of the system in India supplement water from a community borehole with water supplied by privately operated tanker trucks (Miller, 2011; CDD, 2016). Water use for the non-sewered system in Zambia was negligible as dry pit latrines requiring no water are generally used for on-site containment. In contrast, in the study community in India, pour-flush toilets connected to on-site pits are commonly used, thus non-sewered systems require a nominal amount of water for flushing.

Therefore, local water availability and the capacity of related infrastructure to deliver water to users are important considerations when deciding among sanitation approaches. Bangalore, where the India case studies in our analysis are located, is already one of the most water-stressed cities in the world (WRI, 2015; Times of India, 2016). The national water authority in Zambia also lists diminishing water resources due to climate variability as a major challenge in the future (NWASCO, 2017). Furthermore, waterborne sewerage approaches are dependent on infrastructure necessary to deliver water supply. In Zambia, which relies almost entirely on hydropower, several years of regional drought have led to severe electricity deficits, which has limited the capacity of utilities to supply water to customers (ERB, 2015; NWASCO, 2016). Without an adequate and reliable supply of water, sewered conveyance may function poorly and users may be required to find alternative sanitation options as a 'coping' strategy (Laramee et al., 2017).

Water supply may also be an important contributor to lifecycle energy use of sanitation systems. Within this study, particularly high energy use associated with water supply is found for the systems in India, where a low groundwater table and delivery *via* tanker trucks resulted in a high energy-intensity factor for water supply. Overall, water supply accounted for over half of lifecycle energy use for the sewered system and a third of lifecycle energy use for the non-sewered system in India. Moreover, if water losses during distribution are considered, water supply becomes an even greater contributor to lifecycle impacts, potentially increasing the total energy use by 43% for the sewered system and 15% for the non-sewered system in India. Water losses in piped distribution networks has been shown to be an important contributor to lifecycle impacts of urban water systems by others (Friedrich et al., 2009). However, despite the potential importance of accounting for water supply, few LCAs of sanitation systems include water supply within their analysis scope (Corominas et al., 2013).

In contrast to water supply for sewered systems, motorized conveyance for non-sewered systems is not a substantial contributor to lifecycle energy use or GWP impacts within this study. Primary emptying and conveyance was performed manually (e.g. pushcarts) for the non-sewered system in Zambia; thus, although human energy expenditure was accounted for, it is not surprising that minimal impacts are found due to conveyance. Although motorized emptying and conveyance is used for the non-sewered system in India, less than a quarter of lifecycle energy use and minimal GWP impacts are due to conveyance. This is likely because the *per-capita* volume of fecal sludge transported is minimal as mainly solids are transported (underground infiltration of wastewater occurs for on-site systems), which have already undergone substantial degradation prior to conveyance. Within this study, lifecycle energy use is somewhat sensitive to motorized conveyance distance and vehicle energy factor. Although not modelled in this study, energy use may also be influenced by road conditions and traffic congestion. Moreover, these factors may impact labor and financial costs as noted by others (Kennedy-Walker et al., 2015).

4.5.2 Infrastructure embodied energy

Infrastructure embodied energy accounts for a substantial percentage of overall lifecycle energy use for low-tech, passive treatment systems. Within this study, materials, transport and construction required for SSC infrastructure, accounted for approximately 40 – 80% of lifecycle energy use. Treatment infrastructure is the greatest embodied energy contributor for sewered systems, while containment infrastructure is the greatest contributor for non-sewered systems analyzed. All systems within this study rely primarily on passive and biological treatment with minimal regular energy inputs or addition of chemical agents; thus, minimal energy is required for treatment during the use phase. This contrasts with conventional, aerobic wastewater treatment systems, for which energy during the use phase may amount to 75% to 95% of lifecycle energy use due to substantial energy requirements for aeration (Emmerson et al., 1995; Tillman et al., 1998; Stokes and Horvath, 2010; Remy and Jekel, 2011). In fact, many LCAs investigating conventional wastewater treatment plants exclude infrastructure from the analysis and focus solely on the use phase to assess environmental impacts (Corominas et al., 2014).

However, similar to findings from this study, other studies investigating passive, low-tech wastewater treatment systems have found most lifecycle energy use results from infrastructure rather during the use phase (Machado et al., 2008; Kalbar et. al., 2013; Corominas et al., 2014). Thus, future LCAs investigating low-tech sanitation systems should include infrastructure within the analysis scope to understand the full lifecycle energy requirements.

4.5.3 Global warming potential: use phase process emissions

Most of the lifecycle GWP impact occurs during the use phase for passive, anaerobic treatment systems due to biological process emissions. Within this study, approximately 70 – 90% of lifecycle CO_2e emissions result from CH_4 production during anaerobic treatment processes. Nearly all CH_4 emissions occur during the containment phase (92-96%) for non-sewered systems and during the treatment stage for sewered systems (>99%). Thus, interventions to reduce emissions during containment of non-sewered systems and to recover emissions during treatment processes of sewered systems will have the most significant impact on lifecycle GWP.

Despite the potential for substantial GWP contributions from biological treatment processes, Corominas et al. (2014) report that in an analysis of 45 recent wastewater treatment system LCAs, just over half did not assess direct GHG emissions. Instead, LCAs investigating conventional aerobic wastewater treatment plants report that electricity input during the use phase, rather than biological treatment processes, is a more critical source of emissions (Emmerson et al., 1995; Stokes and Horvath, 2010). Aerobic treatment produces primarily CO_2, considered to be biogenic and therefore is generally not included in the overall GHG emission balance. Thus, the extent of GWP during the use phase for conventional, aerobic wastewater treatment plants may be more dependent on the local electricity mix and associated CO_2e emission factors (Friedrich et al., 2007; Gallego et al., 2008).

The greatest contribution of GWP impacts occurs at the containment stage for non-sewered systems, accounting for approximately 90% of impacts. GWP at the containment stage is primarily due to biological processes, calculated according to the extent of COD degradation during storage and the extent to which the pit is aerobic *versus* anaerobic. Therefore, interventions at the containment stage will have the greatest potential to reduce impacts and should focus on minimizing COD degradation, minimizing anaerobic conditions during storage, or minimizing both.

The extent of COD degradation for pits in India and Zambia is based on data by Nwaneri et al. (2008) and the assumption that emptying occurs infrequently (e.g. several years between emptying events). The extent to which COD degrades under aerobic *versus* anaerobic conditions, and thus the extent to which CO_2 *versus* CH_4 is produced is based on an assumed MCF assigned according to field conditions (IPCC, 2006). Pits for the non-sewered system in India were assigned a greater MCF factor (0.7) due to the use of flush water, while a lower MCF was assigned in Zambia (0.5) for dry pits with greater than 5 users. Thus, pit storage conditions in India are assumed to be more anaerobic and thus will emit a greater proportion of CH_4 resulting from COD degradation. If storage conditions are instead assumed to be primarily dry (aerobic) – for example, as might be the condition for a well-functioning urine diversion toilet – or if regular removal of sediment is assumed – for example, as might occur with container based sanitation, IPCC instead recommends a MCF of 0.1 (IPCC, 2006). Under these conditions, the overall lifecycle GWP would be reduced by 64% and 70% from baseline conditions for the non-sewered system in India and Zambia, respectively. However, the greater requirements for conveyance, including emptying frequency and volume of waste transported, as well as additional cost for labor would need to be considered when comparing such scenarios.

The greatest uncertainty for the non-sewered system also occurs at the containment stage, primarily due to the uncertainty range recommended by IPCC (+/- 50%). Empirical data on the extent of aerobic *versus* anaerobic conditions, and thus the extent of CH_4 emissions, under varying use and soil conditions would help to reduce uncertainty associated with lifecycle GWP.

Biogas recovery is a major driver of energy and GWP impacts for decentralized sewered systems utilizing anaerobic treatment processes. When no biogas is recovered, CH_4 produced during anaerobic treatment is released directly as GHG emissions and no reduction in energy use is achieved *via* offsets of conventional fuels. The sewered systems have a substantially higher energy use (3 to 7 times higher) and GWP (2 times higher) impact *per capita* compared to the non-sewered systems when no biogas is recovered. Little to no biogas recovery may occur, for example, if (1) insufficient demand for biogas exists in proximity to biogas supply, (2) infrastructure to facilitate use – i.e. gas enclosure, pipelines and gas appliances – is poorly maintained, which has been observed for some 50% of biogas installations across Asia and sub-Saharan Africa (Bond and Templeton, 2011); or (3) no provision for biogas recovery from anaerobic treatment exists, for example *via* use of conventional septic tanks (IPCC, 2006; Leverenz et al., 2010). On the other hand, at high percentages of biogas recovery, the sewered systems may have lower energy use and GWP impacts relative to the non-sewered systems, particularly when biogas offsets solid fuel use. At full biogas recovery, energy use and GWP impacts may be reduced by an estimated 131% and 105% in Zambia, respectively, and 71% and 67% in India, respectively. Thus, in Zambia, due to the offset of charcoal, net-positive energy and net negative GHG emissions are possible at high percentages of biogas recovery.

In contrast, the percentage of biogas recovery had comparatively less influence on lifecycle energy use or GWP impacts for non-sewered systems. Substantial waste degradation is assumed to take place during long storage periods in pit latrines. Therefore, most biogas is released during the containment phase, where no mechanism to capture and reuse biogas is available. Overall, energy and GWP impacts were reduced by 50% and 11% for the Zambia non-sewered system, respectively, and by 2% and <1% for the India non-sewered system, respectively. Greater energy recovery could be achieved with more frequent collection of waste. However, this benefit would need to be weighed against the additional costs of greater fuel use and labor for emptying and conveyance of waste. Additionally, infrastructure requirements and costs for the fecal sludge treatment plant would increase as fresh, rather than partially degraded and stabilized, waste would be collected and require treatment.

Energy could also be recovered in the form of solid fuels. Only biogas recovery is considered in this analysis as no other forms of energy recovery were reported for any of the case study systems. However, reuse of biosolids could be another form of energy recovery, particularly for the non-sewered systems. Assuming a calorific value of 10.9 – 13.4 MJ/kg dry matter for sludge from drying beds, a further 48 – 59 MJ *per capita* per year could be recovered for each non-sewered system based on the kg of dry solids input into each system *per* day (Gold et al., 2017). This amounts to a 212% and 67% reduction in *per-capita* energy use from baseline levels. Higher *per-capita* energy from solids could theoretically be recovered from fresh feces, *versus* partially stabilized sludge, as all original carbon would be available (Muspratt et al., 2014). Additionally, aerobically stabilized feces would also theoretically yield greater *per-capita* biomass as solid fuel energy relative to the primarily anaerobic degradation in this study. However, trade-offs with more frequent collection for fresh feces and conveyance of greater volumes of waste for aerobic stabilization would need to be considered.

Baseline fuel type also drives the magnitude of energy use and GWP reductions associated with biogas recovery, but also influences the uncertainty of impacts. The model developed in this study takes relative fuel efficiencies into consideration, thus greater energy savings are estimated when inefficient fuels, such as charcoal, are offset by use of biogas (Kaoma and Kasali, 1994; Anozie et al., 2007; Laramee et al., 2017). Furthermore, greater GWP reductions are modeled for charcoal *versus* propane offset due to the greater GHG emissions associated with charcoal production and use compared to relatively clean burning propane use (Smith, 1994; Kammen and Lew, 2005). Thus, within this study, greater impact reductions are modeled for both the sewered and non-sewered systems in Zambia *versus* India due to baseline use of charcoal *versus* propane, respectively. However, baseline fuel use also influences the uncertainty of energy use and GWP impacts, which were found to be most sensitive to variables which influence the volume of biogas recovered from the primary treatment unit. The product of these variables is multiplied by the efficiency ratio of baseline fuel to biogas to determine the total energy offset. Thus, when a comparatively inefficient fuel such as charcoal is offset, the uncertainty range is multiplied according to a relatively large ratio resulting in a large uncertainty range.

Nutrients contained in urine and excreta are another potential resource that have not been considered in the present analysis. In theory, the nitrogen and phosphorus contained in excreta could potentially be recovered and reused for agricultural purposes and could offset synthetic fertilizer use. In particular, the production of ammonia, used for synthetic nitrogen fertilizer production, is an energy intensive process which accounts for an estimated 3 - 5% of global carbon emissions (Scientific American, 2013). If all nitrogen produced per capita was recovered, and if reuse of this nitrogen offset synthetic fertilizer use, 207MJ and 226MJ per capita of energy could theoretically be offset annually in Zambia and India, respectively, while annual emissions could be offset by 18kgCO_2e to 20kgCO_2e per capita, respectively.[59] This would amount to a 74 – 96% and 282 – 486% reduction in *per-capita* energy use and a 19 – 30% and 23 – 24% reduction in *per-capita* CO_2e emissions for sewered and non-sewered systems, respectively. However, in practice, wastewater and bio-solids are typically reused for only small-scale and informal agricultural applications where synthetic fertilizers are not typically used. Therefore, the reuse of nutrients in such setting would likely not offset synthetic fertilizer production. Additionally, the energy and emission cost of converting nitrogen into a safe form for reuse has not been accounted for in this approximate calculation.

4.5.4 Lifecycle cost analysis

Within this study, user and agency economic costs are higher for sewered systems *versus* non-sewered system in both countries. Sewered system user costs are primarily driven by water supply and most agency costs are due to capital investment for infrastructure. Notably, the present analysis does not include land costs due to a lack of available data. Within this study, the *per-capita* land area for sewered systems is approximately 12 times greater than for non-sewered systems. Thus, the magnitude of cost difference between sewered *versus* non-sewered systems is likely even higher than our findings indicate.

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[59] Assuming nitrogen production of 8.8g per capita per day in Zambia and 9.6 g per capita per day in India (FAOSTAT, 2013) and energy and emission factors for production of synthetic fertilizers of 64.1 MJ / kg N and 5.7 kgCO2e/ kg N,

Notably, an opposite trend in cost distribution between capital and operational expenditures for sewered *versus* non-sewered systems is found. The agency assumes nearly all capital costs for the sewered systems, as most costs are incurred for communal sewer pipes and treatment infrastructure. In contrast, non-sewered system users shoulder much of the initial cost as the user (e.g. household or landlord) is generally responsible for investment in on-site containment infrastructure. Non-sewered communal treatment infrastructure is a comparatively small *per-capita* cost. Therefore, from an initial cost perspective, investment in sewered system approaches are likely to be preferred by the user, whereas non-sewered sanitation approaches are likely to be favored by the agency. This also suggests that where policies are in place to subsidize communal sanitation infrastructure, an equitable policy would be to also subsidize investment in, or improvements to, on-site sanitation.

Conversely, operating costs are high for sewered system users, primarily due to the use of water required for waste conveyance; whereas operating costs are relatively low for the sewered system agencies. The opposite trend is found for non-sewered systems: operating costs are low for the user, particularly for the non-sewered system in Zambia for which no water is required for operations, and relatively high for the agency due to the labor intensity of pit emptying. The proportion of labor costs are particularly high in Zambia, where a high volume of solid waste in pits impedes emptying, accessibility for emptying at the pit structure is minimal, and waste is transported manually due to dense housing and poor road conditions.[60] Thus, optimizing the speed of pit emptying, including deterring solid waste disposal in pits and improved accessibility would help to reduce operating costs for non-sewered system agencies, which would also bring emptying costs down for the user.

Recovered wastewater could be an important source of revenue for the sewered systems. However, no systems analyzed systematically recovered water at the time of this study.

[60] The agency pit-emptying team will typically break an opening through the masonry pit substructure to gain access to waste for emptying. The agency is then responsible for the time, materials and cost of repairing the pit sub-structure.

Assuming recovered wastewater was priced equivalently to potable water supply, annual agency costs could be reduced by 25% for the sewered system in Zambia and by 44% for the sewered system in India. However, it is unlikely that users would be willing to pay the same price for wastewater and potable water. Additionally, post-treatment is likely necessary as pathogen levels in DEWATS effluent may be above recommended levels for reuse (Foxon et al., 2004).

Despite the potential for biogas recovery to offer considerable energy and GWP benefits, this study finds minimal potential for financial benefits of biogas recovery relative to the total capital and operating costs of sanitation systems. Carbon credits also offer negligible financial benefits under current carbon prices. Thus, at current energy and carbon pricing, biogas recovery will do little to financially incentivize capital and recurrent costs of sanitation services. However, the current prices of energy in this study are not reflective of actual costs to produce and supply energy in both countries due to subsidies on electricity and propane in Zambia and India, respectively. If energy prices reflected actual costs, energy recovery benefits may be up to three times higher than currently modeled in Zambia and 1.5 times higher in India (Times of Zambia, 2016; Indian Express, 2016). Moreover, carbon credits could be a meaningful source of revenue for sewered systems at the upper bound of pricing considered. However, macro-economic mechanisms to ensure pricing at this level are necessary to access such benefits.

The additional investment required only to facilitate biogas recovery from sanitation infrastructure could, however, be economically favorable. For example, within this study, the equivalent annual cost for additional infrastructure and equipment required specifically for biogas supply (e.g. piping and stoves) is positive for all systems except the non-sewered system in India where very little biogas is recovered. Thus, while biogas energy may not contribute substantially to the cost of providing sanitation services, the cost of biogas infrastructure could be recouped *via* energy provision. Such an investment would also enable positive energy and GWP benefits. Recovery of bio-solids for reuse as solid fuels could provide additional economic benefits. Using the cost of conventional fuels as a proxy, and taking relative fuel efficiencies into consideration, solid fuel recovery could offer economic benefits on the order of 5 – 10% of total costs.

4.6 Conclusions

4.6.1 Implications

By 2015, 44% and 37% of the urban population in Zambia and India lacked access to improved sanitation. Urban populations in these countries are expected to grow by 1.9 and 1.4 times, respectively. Thus, an estimated 8.5 million people in urban Zambia and 318 million people in urban India will require new sanitation infrastructure by 2030 to meet the SDG target of universal access to sanitation (WHO/UNICEF, 2015a; UN-DESA, 2015). Moreover, the population requiring safe treatment and disposal of waste is higher than these figures suggest as the sanitation access percentages reported by WHO/UNICEF (2015a) are based only on access to a toilet facility (Baum et al., 2014).

Findings from this study suggest that to provide sanitation for this unserved population by 2030, the choice amongst infrastructure and energy recovery alternatives analyzed within this study could result in annual differences on the order of 0.2 billion cubic meters of water (15% of total annual withdrawals), 2,647 TJ of energy (1.3% of total energy use) and 1.2 Mt CO_2e (7% of total GHG emissions) in Zambia; and 2.5 billion cubic meters of water (0.3% of total annual withdrawals), 44,555 TJ of energy (0.6% of total energy use) and 23.1 Mt CO_2e (0.9% of total GHG emissions) in India.[61] Moreover, the total difference in costs incurred by public utilities depending on sanitation infrastructure choice could be on the order of US $1 billion (1.6% of GDP) in Zambia and US $9.3 billion (0.1% of GDP) in India.[62] Finally, user costs range from 0.4 – 1.9% of gross national income (GNI) *per capita* in Zambia and 0.3 – 0.4% of GNI *per capita* in India.[63]

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[61] Water use data from World Bank (2014) databank, indicator: "Annual freshwater withdrawals, total (billion cubic meters)" (http://data.worldbank.org); Residential energy use (country level) data from International Energy Association (2014) (https://www.iea.org/sankey/); GHG emission (country level) data from WRI CAIT (2013) (http://cait.wri.org/).

[62] GDP *per* country from World Bank (2015) databank, indicator: "GDP, PPP (current international $)" (http://data.worldbank.org).

[63] GNI *per capita* from World Bank (2015) databank, indicator: "GNI *per capita*, PPP (current international $)" (http://data.worldbank.org).

4.6.2 Limitations

The WHO/UNICEF Joint Monitoring Programme indicator for 'safely managed sanitation' is used to define the functional unit of the present analysis. However, this indicator does not offer specific criteria on how 'safe management' is defined, and thus systems are compared which may have varying levels of effectiveness in regard to 'safe' management. Specifically, no treatment performance criteria are specified, such as minimum discharge standards for either effluent or sludge. Additionally, on-site management of waste, described by the Joint Monitoring Programme as 'separation of excreta from human contact including either safe disposal *in-situ* or transport and treatment offsite' does not address complete containment of liquid wastes. (WHO/UNICEF, 2017). For example, on-site pit latrines are rarely lined completely and septic tanks or leech pits also allow for infiltration of partially treated or untreated wastewater. In each scenario, liquid waste may enter the surrounding environment posing groundwater contamination and public health risks (Graham and Polizzotto, 2013; Nyenje et al., 2009). However, despite limitations of the JMP sanitation indicator, this study was limited by insufficient data to more accurately assess the extent to which case studies analyzed provide 'safely managed sanitation.' Ultimately, the sanitation sector is also limited with regards to what constitutes 'safe' management in terms of the relative public health outcomes of various types of sanitation infrastructure (Wolf et al., 2014).

Energy and GWP findings for non-sewered systems are limited by dependency on two uncertain and data-poor parameters; COD degradation during containment and an assumed latrine MCF. Due to a lack of available data, estimates of COD removal during containment are based on limited data from ventilated improved pit (VIP) latrines in South Africa (Nwaneri et al., 2008). Different COD removal rates likely exist in other areas depending on ground conditions, lining of pits, water use and moisture content, temperature, etc. MCFs were assigned based on IPCC (2006) guidelines and general pit latrine conditions in each study site. However, IPCC recommended MCFs vary widely from 0.1 to 0.7 with an uncertainty range of +/-50% depending on ground conditions, number of users and presence of flush water (IPCC, 2006). Further data collection characterizing pits and the degree of aerobic *versus* anaerobic degradation depending on site and use conditions would help to verify and reduce uncertainties on COD removal rates and methane production.

Moreover, the approach used to calculate the number of people served by non-sewered systems is also limited by available data. Estimates are based on daily *per-capita* production of excreta reported by Rose et al. (2016) and organic solids content of fresh and degraded excreta reported by Nwaneri et al. (2008); each study is based on a limited sample number (*n=8* studies and *n=16* latrines, respectively). The method proposed in this study additionally assumes a constant organic material fill rate between pit emptying events, which is unlikely due to long periods between emptying and fluctuating populations.

Due to a lack of available data, this study has relied on international databases for material and transportation energy and emissions factors, rather than using country specific data. Moreover, limited data is available on *per-capita* production rates including COD, water use, wastewater generation, etc. This study, and future LCAs in developing country settings, would benefit from greater availability of country specific data. The large uncertainties associated with our findings could additionally be reduced with more data availability.

Finally, findings from this study are also limited to only one sewered and one non-sewered system in specific locations in Zambia and India. While this study highlights potential drivers of impacts for each approach, several arrangements of each system type exist with varying parameters including infrastructure requirements, materials, energy inputs and labor requirements. Moreover, the magnitude of costs and benefits are dependent on geographical, social and institutional context, and thus will vary depending on location. Further studies on varying system arrangements and in different geographies are needed to make more generalized statements on the costs and benefits of sewered and non-sewered systems.

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5 Conclusion

The 2015 Sustainable Development Goals (SDGs) established an international pledge to "end poverty, protect the planet and ensure prosperity for all" including achieving universal access to safely managed sanitation and modern, clean fuels (UN, 2015). The SDGs and the 2015 Paris Agreement on Climate Change additionally set a global agenda to integrate climate change adaptation and mitigation measures into national policies, strategies and planning including fostering low emissions development (UN, 2015). Major infrastructure investments will be needed to meet sanitation and energy SDG targets, particularly for underserved and rapidly growing populations in sub-Saharan Africa (SSA) and South Asia (SA). This infrastructure challenge is compounded by increasingly scarce resources and variable climatic conditions. At the same time, these challenges also pose an opportunity for emerging cities to develop smart, resource-efficient urban systems and avoid investments that would lock cities in to costly and resource intensive infrastructure for decades to come (Beard et al., 2016).

In response to these development and resource challenges, integrated sanitation and energy recovery strategies have received increasing attention to offset operational costs of sanitation services, provide alternative energy resources and reduce environmental impacts of waste treatment systems. Moreover, new sanitation management approaches – including decentralized sewered and non-sewered options – have gained traction in low-income SSA and SA cities as alternatives to expensive and resource intensive conventional wastewater treatment. However, limited empirical evidence is available on the costs and benefits of such alternatives and, moreover, no standard approach exists to quantitatively evaluate and compare new sanitation options.

The research in this dissertation sought to contribute knowledge to this gap in the practice of sanitation planning by (1) providing empirical evidence on integrated sanitation and resource recovery strategies and (2) developing and demonstrating a quantitative lifecycle framework to evaluate resource, environmental and economic costs and benefits of alternative sanitation approaches including modeling the extent to which resource recovery impacts costs and benefits.

This chapter summarizes the main conclusions and contributions from Chapters 2, 3 and 4, discusses implications for sanitation planning and provides recommendations for future work.

5.1 Contributions

Key contributions from each chapter of this dissertation are discussed below.

Chapter 2 contributes empirical evidence on small-scale anaerobic digesters (ADs) used in rural, agricultural settings at the household level in Northern Tanzania. A cross-sectional study of 40 households suggests that in this context – i.e. in temperate climates with domestic dairy farming and common use of solid wood fuels – domestic AD adoption is associated with lower fuel-wood use, energy-related expenditures, time-costs of energy procurement, and CO_2e emissions, as well as with higher farm incomes. Moreover, findings from this study suggest that ADs in this context are a favorable financial investment under a wide range of discount rates and offer considerable financial savings to the household over the lifespan of the system. Previous to this study, limited evidence was available regarding the existence or magnitude of socioeconomic or environmental benefits accruing from AD implementation in SSA despite substantial programmatic investment.

Chapter 3 contributes empirical evidence on communal ADs used for treatment of domestic wastewater using primary data collected from 15 AD installations in 3 peri-urban communities in Zambia. Although communal ADs have been promoted as a waste-to-energy strategy to improve the environmental and financial sustainability of sanitation services, little evidence is available regarding the performance of such systems in field conditions. Findings from this study suggest that treatment efficiencies of ADs are comparable to, or greater than, efficiencies of conventionally used septic tanks (STs). Moreover, ADs used in this context offer considerable GHG emission benefits, amounting to a reduction of up to 12% of total *per-capita* emissions in Zambia relative to STs. This study also identifies the conditions under which ADs are likely to perform well – namely, in locations with adequate water access, with common use of solid fuels (e.g. charcoal) and where biogas supply and demand can be spatially optimized. Finally, this study identifies the financial criteria necessary for favorable investment in biogas recovery: under the conditions identified for favorable performance of ADs, a minimum biogas tariff of US$0.26/$m^3$ methane – competitive with conventional fuels used – or a minimum carbon price of US$10 *per* tCO_2e would be necessary to cover the additional cost of facilitating biogas recovery from primary wastewater treatment systems.

Chapter 4 contributes to the knowledge and practice of urban sanitation planning by developing a lifecycle framework that allows decision-makers to comprehensively evaluate the water and energy use, GHG emissions and financial costs and benefits of sanitation infrastructure alternatives, including the extent to which resource recovery affects these costs and benefits. Application of this framework is demonstrated using primary and secondary data collected from two decentralized sewered and two non-sewered sanitation systems located in Zambia and India, thus also generating the first comparative analysis of these sanitation management approaches in urban SSA and SA.

Through analysis of these case studies, Chapter 4 identifies key potential drivers of impacts both within and between sewered and non-sewered approaches. In the context of the case studies analyzed, (1) capital infrastructure and water supply were identified as key drivers of lifecycle energy use and (2) direct methane emissions were identified as the primary driver of GWP impacts. Additionally, the percent of biogas recovery and baseline fuel use were found to strongly influence lifecycle energy use and GWP impacts or sewered systems, whereas these factors had little influence for non-sewered systems. Findings from this study also suggest significantly higher user and agency costs for sewered *versus* non-sewered systems; however, non-sewered system users may incur higher upfront capital costs and a greater proportion of lifecycle costs compared to sewered system users.

5.2 Implications

Key implications for sanitation planning that can be drawn this dissertation include:

Integrated sanitation and energy recovery approaches can offer significant energy and GWP benefits. Findings from this dissertation suggest that biogas recovery from domestic wastes has the potential to offer significant energy and GHG emission benefits. Adoption of domestic ADs in agricultural settings in Tanzania was found to significantly reduce fuel-wood use and lower CO_2e emissions relative to households without ADs (Chapter 2). Use of communal ADs for domestic wastewater treatment in peri-urban Zambia was found to reduce annual *per-capita* GHG emissions relative to conventional STs (Chapter 3). Considering the conditions under which ADs are likely to perform well, as identified in this study, will help sanitation planners optimize benefits from communal ADs. Finally, in Chapter 4, the percentage of biogas recovery was found to strongly influence the lifecycle energy and GWP impacts of decentralized sewered systems utilizing anaerobic treatment processes. Depending on baseline fuel use, findings from this dissertation show biogas recovery has the potential to achieve a net energy savings and a net reduction in GHG emissions over the lifespan of sewered systems with anaerobic treatment, even including the materials required to construct systems. Although biogas recovery did not strongly influence the non-sewered systems analyzed (where infrequent collection results in a loss of biogas during the containment stage), other forms of energy recovery such as solid fuel recovery show promise for providing considerable energy and GWP benefits.

Greater economic incentives and different models are required to attract investment in integrated sanitation and energy approaches. Despite the compelling evidence on energy and GHG emission benefits, the financial benefits of the waste-to-energy approaches investigated in this dissertation have mixed results. Household-scale ADs using agricultural feedstocks (e.g. dairy cattle manure) as a feed material were shown in Chapter 2 to be a financially attractive investment with a simple pay-back period of less than 4 years. These systems differ from the ADs investigated in Chapter 3 in that (1) they utilize manure as a feedstock (rather than only domestic wastewater), which has the potential to produce a far greater volume of methane and (2) the primary objective of domestic, agricultural installations is energy production (rather than wastewater treatment). However, despite the apparent favorability of this investment, few low-to-middle income households in SSA have the financial liquidity to pay for such an investment. Given the clear energy and financial benefits shown by this study, microfinance loans could be a possible option to catalyze investment in domestic, agricultural systems. However, further research would be necessary to understand the effectiveness of this approach.

In contrast, a different financial model is necessary for the communal ADs investigated in Chapter 3, which are operated by water and sewerage utilities rather than by households. Findings from this chapter suggest that a biogas tariff set at a price competitive with conventional fuels could offset the additional cost required to facilitate biogas recovery from primary wastewater treatment. Although the cost-competitiveness of this tariff shows promise, further studies are necessary to investigate household willingness to pay for biogas. Additionally, the minimum carbon price necessary to offset the additional cost of facilitating biogas recovery (US \$10 *per* tCO_2e) and, moreover, the price required to offset the total capital and recurrent costs of primary wastewater treatment (US \$46 *per* tCO_2e) are both far higher than the current market price of carbon. However, these prices are within the range of carbon prices currently discussed – e.g. on the order of the 'social cost of carbon' used by the US-EPA (US \$37 *per* tCO_2e) for cost-benefit analyses and less than the price called for by the United Nations Global Compact (US \$100 *per* tCO_2e). Although the SDGs call for 'low emissions development,' and communal ADs are shown in this dissertation to have promising sanitation, clean energy and climate co-benefits, a stable carbon market with pricing on par with these levels is necessary to incentivize investment in emissions reduction projects.

In Chapter 4, potential revenue from biogas recovery was found to be only a minimal percentage of the overall costs of sanitation provision (less than 9% for sewered systems, less than 3% for non-sewered systems). At currently available carbon prices, potential revenue generated *via* carbon offsets would also be minimal (less than 1% of overall costs of sanitation provision for all systems). Carbon financing could potentially be a meaningful source of revenue for sewered systems at the high range of future carbon prices discussed (\$100 *per* tCO_2e), offsetting 18 – 29% of the annual cost of sanitation provision. However, currently, this price of carbon is only theoretical. Other forms of resource recovery may be more lucrative and should be explored – for example, water recovery for decentralized sewered systems and bio-solids fuel recovery for non-sewered systems.

The case studies analyzed also highlight the different absolute costs and cost drivers between systems, as well as the differing distribution of costs between user and agency – all of which have implications for financing of sanitation services. Decentralized sewered costs were higher for both the user and agency compared to the non-sewered systems for both Zambia and India case studies. Thus, from a lifecycle financial perspective, non-sewered options are likely to be more favorable, particularly for governments and municipalities financing sanitation infrastructure and services. However, due to the high cost of on-site containment, users pay higher upfront capital costs for non-sewered *versus* sewered systems, which may be a significant obstacle for low-income users to access improved sanitation. Although the upfront capital costs are high for non-sewered systems, the operational costs are low. This would suggest that a scheme in which the upfront cost is subsidized and then recouped through a higher tariff for operational costs could increase access to sanitation.

Overall, lifecycle user costs were driven by water supply costs for both sewered systems and for the non-sewered system in India, whereas the capital cost of on-site sanitation was the main cost driver for users of the non-sewered system in Zambia. Agency costs were driven by capital costs for sewered systems and labor costs for non-sewered systems. Thus, capital and operational costs are inversely distributed between users and agencies for sewered *versus* non-sewered systems suggesting different financing models are needed for different sanitation approaches.

Use of a system-wide lifecycle assessment (LCA) and lifecycle cost (LCC) framework provides key insights on resource use, GWP impacts and economic costs and benefits of sanitation approaches. By developing and applying a lifecycle framework to four case studies across different geographies, cultural contexts and institutional arrangements, this dissertation provides key insights on the potential drivers of resource use, environmental impacts and financial costs of sewered and non-sewered sanitation approaches. Although these findings are specific to the case study settings and sanitation approaches analyzed, identifying key drivers nevertheless provides guidance for sanitation planners on (1) identifying where the greatest reductions in resource use and environmental impacts are likely to be made for different sanitation approaches and (2) which system features and processes are critical to include within the scope of future lifecycle assessments.

The key drivers identified in this study also demonstrate that a system-wide approach, rather than simply a process approach, is essential in understanding and comparing the full lifecycle costs and benefits of different sanitation alternatives. For example, within the case studies analyzed in Chapter 4, including energy required for water supply and the effects of biogas recovery on conventional energy use were essential to understanding the lifecycle energy and GWP of the sanitation alternatives considered. This system-wide lifecycle approach stands in contrast to most wastewater related LCAs, which generally investigate specific processes or technologies within a larger system (Corominas et al., 2013). While comparisons between technologies may be, for example, helpful for engineering decisions to minimize impacts of a specific treatment process, a system-wide approach is necessary for urban planning decisions – for example whether a sewered or non-sewered approach is the 'best' solution within the resource or financial constraints of a specific context. This interpretation is in line with the few LCAs that have also considered expanded system boundaries (Lundin et al., 2000; Lundie et al., 2004).

Urban sanitation planning in low-income country contexts does not typically include assessing the lifecycle resource and environmental costs of sanitation alternatives. However, more careful consideration of resources required for operation of various sanitation approaches will be increasingly important in rapidly expanding and resource constrained contexts. For example, consideration of local water stress is critical in contexts like the India case study settings where, due to diminishing water resources, water supply was both the greatest energy contributor and the highest cost for users of the decentralized sewered system. Through developing and applying a lifecycle framework to analyze four alternative sanitation approaches, this dissertation demonstrates that key insights on resource use, environmental impacts and financial costs can be gained through use of this methodology. The framework developed in this dissertation can be used to evaluate other alternative sanitation approaches currently being developed and implemented to better understand long-term resource and environmental impacts.

5.3 Future research

This dissertation investigates only a limited number of waste-to-energy approaches and, moreover, only limited dimensions of these approaches. Several potential research efforts that would build on the contributions in this dissertation are discussed below:

Investigate why so many anaerobic digesters in low-income countries fail (and develop financing incentives to ensure greater numbers succeed). Findings from this dissertation suggest that significant energy and GHG emission benefits can be gained from operational ADs. Results from Chapter 2 also indicate that significant financial savings may be achieved through use of household ADs in rural, agricultural settings. Yet, notoriously high rates of failure are reported for ADs throughout the developing world (Bond and Templeton, 2011). Further research investigating why so many systems fail, and the conditions under which ADs perform well, would be helpful in efforts to exploit the full benefits ADs offer. Findings from this dissertation suggest that the greatest benefits from ADs treating communal wastewater will be realized under conditions with adequate water access, use of low-efficiency fuels and adequate demand for biogas in proximity to supply. Further research to investigate the generalizability of these findings would help to inform future investment in AD implementation. Additionally, without sufficient financial incentives, operators may have little motivation to maintain biogas recovery infrastructure. Findings from Chapter 3 show that the additional cost required to facilitate biogas recovery from anaerobic wastewater treatment systems could potentially be paid for in less than 5 years *via* a biogas tariff equivalent to what households with biogas supply are currently saving. However, establishing a pricing mechanism is necessary to take advantage of this potential financial benefit.

Investigate COD removal and methane emissions from pit latrines under varying site conditions. Methane emissions from pit latrines were identified in Chapter 4 as a key driver of GWP impacts of non-sewered systems. However, these findings were based on two uncertain and data-poor parameters: COD degradation during containment and an assumed latrine methane correction factor (MCF). Due to a lack of available data, COD removal estimates during containment are based on limited data from ventilated improved pit (VIP) latrines in South Africa (Nwaneri et al., 2008). Different COD removal rates likely exist in other areas depending on ground conditions, lining of pits, water use and moisture content, temperature, etc. MCFs were assigned based on IPCC (2006) guidelines and general pit latrine conditions in each study site. However, IPCC recommended MCFs vary widely from 0.1 to 0.7 with an uncertainty range of +/-50% depending on ground conditions, number of users and presence of flush water (IPCC, 2006). Further data collection characterizing pits and the degree of aerobic *versus* anaerobic degradation depending on site and use conditions would help to verify findings and reduce uncertainties on COD removal rates and methane production.

Investigate decentralized sewered and non-sewered sanitation and resource recovery approaches across a range of different contexts to improve generalizability and identify optimal strategies. Several sanitation and resource recovery strategies already exist, and many new approaches are emerging in response to increasing stress on already limited resources and global pledges to minimize environmental impacts. This dissertation only investigates a limited number of approaches in a limited number of geographies. The magnitude of costs and benefits of sanitation approaches are likely to be shaped by contextual socio-economic, physical and institutional factors. The generalizability of findings from this dissertation would be strengthened by investigating sewered and non-sewered systems across a range of different contexts.

Moreover, the framework developed in this study could be applied to new sanitation options to understand the benefits and tradeoffs of different approaches. For example, more advanced wastewater treatment methods may achieve higher effluent quality, thus creating more opportunities for wastewater reuse. Although greater energy inputs may be required to achieve higher standards, energy may also be offset through water reuse. These tradeoffs could be weighed to understand net costs or benefits. Mechanized dewatering technologies could be used to accelerate fecal sludge treatment, thus increasing system capacity and potentially generating more revenue. However, this would need to be weighed against increased energy costs. Additionally, container-based sanitation may reduce containment-stage emissions and may provide opportunities for greater resource recovery. However, greater treatment infrastructure may be required for treating fresh waste and more frequent collection may increase energy use, GHG emissions and labor costs. Finally, the scale at which various sanitation approaches may be relatively more beneficial is an important consideration that could be investigated.

Additionally, different resource recovery options could be explored. In this dissertation, biogas recovery as a cooking fuel replacement is investigated. However, several other energy, nutrient and water recovery options are possible and may be achieve greater benefits depending on the system and context. For example, estimates in Chapter 4 indicate that bio-solids recovery could net 4 – 20 times greater financial benefits compared to biogas recovery. Moreover, water supply was identified in Chapter 4 as the highest cost factor for operations of the sewered systems, particularly for the system in India where water costs were especially high due to limited resources and supply *via* tanker trucks. Thus, water recovery may be a key source of financial revenue and may also reduce lifecycle energy use by offsetting supply. However, willingness to pay for treated wastewater would need to be examined and higher effluent standards would need to be achieved for most reuse options.

Expand lifecycle framework to include other parameters such as nutrient flows, health impacts and social aspects. The current lifecycle framework does not consider the eutrophication impact of nutrient release from waste streams and additionally, does not include the potential benefits of nutrient recovery and replacement of synthetic fertilizers. Discharge of nutrients, such as nitrogen and phosphorus, contained in wastewater and fecal sludge can have substantial negative environmental impacts. On the other hand, recovering nutrients could also have potential environmental and financial benefits by offsetting conventional fertilizer use (Tillman et al., 1998; Asano et al., 2007). The framework developed in this dissertation could be expanded to include nutrient flows. Nutrient input, removal and discharge from sewered systems would be relatively straightforward to assess. However, non-sewered systems would need to consider nutrient infiltration from on-site sanitation, which will likely depend on site-specific factors such as soil conditions and height of groundwater table, and therefore may be more variable and complex to model (Jacks et al., 1998; Nyenje, 2009; Graham and Polizzotto, 2013).

The framework would also be strengthened by the inclusion of a health impact category. However, little evidence currently exists regarding the relative health outcomes associated with sewered and non-sewered approaches. Best current estimates suggest a 16% risk reduction of incidence of diarrhea in children under-five achieved by upgrading from an unimproved to a non-sewered improved sanitation facility, and a 69% risk reduction achieved by upgrading from an unimproved to sewered sanitation (Wolf et al., 2014). However, these estimates, particularly for sewered connections, should be treated with caution as the figures are based on only a limited number of studies (Wolf et al., 2014). Furthermore, it is generally agreed within the sanitation sector that public health outcomes may not be achieved unless a certain threshold level of community sanitation coverage is achieved, although limited quantitative evidence currently exists defining this threshold (Pruss-Ustun et al., 2014). Various research efforts are currently underway attempting to better characterize the relationship between sanitation interventions, exposure risks, and health outcomes (e.g. SaniPath). As further evidence becomes available, a health impact category could be included in future models.

Other important aspects of sanitation planning include social and institutional and factors. For example, the user acceptance and service level of varying sanitation approaches should be considered; users may perceive waterborne sewerage as ‘modern’ and aspirational, whereas non-sewered approaches may be less desirable. The technical capacity of local operators and institutions should also be considered. Other urban sanitation planning frameworks include considerations of social and institutional aspects (e.g. Sanitation 21; CLUES, 2011). The present framework could be improved through better integration with such models.

Integrate a weighting system to quantify composite endpoint categories. This study reports midpoint impacts rather than endpoint impact categories (e.g. natural environment, human health, etc.). Endpoint impacts may be easier for decision makers to understand. Weighting is generally done by local stakeholders and has not been included in the scope of the current analysis. Endpoint categories can also introduce greater uncertainty. Nevertheless, this framework could be improved as a decision-making tool by integrating a quantitative weighting system. Composite scores could be calculated based on the objectives, needs and constraints in a specific context.

Appendix A: Supporting Information for Chapter 3

Table A.1: Cooking fuel emission factors and global warming potentials (GWP)

Parameter	**CO_2 (CO_2e)**	**CO**	**CH_4**
Emissions *per* unit mass of fuel combusted (g/kg)			
Charcoal (use) (g/kg) [a]	2400	200	6
Charcoal (production) (g/kg) [a]	2085	160	29
Electricity (hydroelectric facility, tropical reservoir) (g/kWh) [b]	1765	-	-
Global Warming Potential, 100-year [c]	1	3.3	28

[a] Kammen and Lew (2005)

[b] Demarty and Bastien (2011)

[c] IPCC (2012)

Table A.2: Cooking fuel energy contents

Energy type	Energy conversion (MJ)		
	Site A	Site B	Site C
Charcoal (kg) [a]	25.9	31.2	28.6
Electricity (kWh) [b]	3.6	3.6	3.6
Methane (m^3) [c]	31.6	31.9	33.1

[a] Tested by Pollution Research Group Laboratories, University of Kwazulu Natal, Durban, South Africa (2015)

[b] Standard definition

[c] Calculated using methane heat of combustion (55.5MJ/kg) and methane density specific to each site

Table A.3: Cooking fuel prices *per* study site

Energy cost *per* unit [a]	**SITE A**			**SITE B**			**SITE C**		
	n	Mean	(SD)	*n*	Mean	(SD)	*n*	Mean	(SD)
Charcoal (bag<5kg) (USD/kg) [b]	*20*	\$0.24	(\$0.03)	*116*	\$0.27	(\$0.10)	*108*	\$0.22	(\$0.07)
Charcoal (bag>20kg) (USD/kg) [b]	*22*	\$0.18	(\$0.03)	*6*	\$0.16	(\$0.03)	*14*	\$0.14	(\$0.02)
Electricity (USD/kWh) [c]		\$0.04			\$0.02			\$0.02	
Methane (USD/m3) [d]		\$0.00			\$0.00			\$0.00	

[a] Exchange rate: \$1 USD = 7.25 ZMW (May 2015, oanda.com)

[b] Charcoal cost calculated *via* local market and household survey (weight of full charcoal bags and cost recorded)

[c] Electricity tariffs obtained *via* utility bills

[d] Biogas provided free of charge at time of household surveys

Appendix B: Supporting Information for Chapter 4

B.1 Case study assumptions, inputs and outputs for lifecycle model

General assumptions: sewered and non-sewered systems – Zambia and India (all study sites)

Table B.1: Energy and emission factors for construction processes, materials and transportation (all study sites)

Item	**Unit**	**Energy factor (MJ / unit)[a]**	**Emissions factor (kg CO_2 / unit)[a]**	**SimaPro description**
Process				
Site clearance	kg	0.0281	0.00197	Excavator, technology mix, 100kW
Excavate 200mm top soil	kg	0.0281	0.00197	Excavator, technology mix, 100kW
Excavation up to 3m depth	kg	0.0281	0.00197	Excavator, technology mix, 100kW
Ground levelling after excavation	kg	0.0281	0.00197	Excavator, technology mix, 100kW
Backfilling and compacting in layers	kg	0.0281	0.00197	Excavator, technology mix, 100kW
Disposal of surplus soil	tkm	3.66	0.253	Transport, truck 10-20t, EURO5, 80%LF, empty return
Material				
Cement	kg	6.03	1.24	Cement, Portland {RoW} | market for | Alloc Def, S
Sand	kg	59.7	4.24	Sand {RoW}| gravel and sand quarry
Aggregate	kg	57.6	5.09	Gravel, round {RoW}| gravel and sand quarry
Clay fired bricks	kg	2.56	0.249	Brick {RoW} | production | Alloc Def, S
Reinforcing steel	kg	25.2	2.39	Reinforcing steel {RoW}| production | Alloc Def, U
Steel (rolled shapes)	kg	29.8	2.6	Cold rolled sheet, steel, at plant/RNA
Galvanized steel	kg	27.6	2.41	Steel hot dip galvanized (ILCD) blast furnace route
Aluminum	kg	182	14.1	Aluminum, cast, precision sand casting/kg/US
HDPE liner	kg	76.4	1.73	Polyethylene, high density, granulate {RoW}| production|Alloc Def, S
PVC pipe	kg	67.5	2.85	PVC pipe E
PVC (‘Rubber’)	kg	56.2	2.38	Polyvinylchloride resin (S-PVC), suspension polymerisation, production mix, at plant, RER
Chlorine disinfectant	Kg	14	0.969	Sodium hypochlorite, without water, in 15% solution state {GLO}|market for | Alloc Def, S
Textile (cotton)	kg	299	22.9	Textile, woven cotton {GLO}|market for | Alloc Def, S
Transportation				

Container ship, ocean	tkm	0.164	0.0129	Container ship ocean, technology mix, 27,500 dwt pay load capacity RER S
Truck > 20 tonnes	tkm	1.44	0.0992	Transport, truck > 20t, EURO5, 80%LF, empty return
Truck 10-20 tonnes	tkm	3.66	0.253	Transport, truck 10-20t, EURO5, 80%LF, empty return
Truck < 10 tonnes	tkm	5.35	0.369	Transport, truck < 10t, EURO5, 80%LF, empty return
Passenger vehicle, large size, diesel	km	5.77	0.381	Transport, passenger car, large size, diesel, EURO 5
Pumping / desludging	m3	19.4	1.35	Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S

[a]All values from SimaPro LCA software unless noted otherwise

Table B.2: Conventional cooking fuel emission factors and global warming potential (GWP) (all study sites)

Parameter	CO_2	CO	CH_4
Emissions *per* unit of fuel combusted			
Charcoal (use) (g/kg) [a]	2400	200	6
Charcoal (production) (g/kg) [a]	2085	160	29
Liquid Petroleum Gas (LPG) (g/kg) [b]	3190	-	0.01
Global Warming Potential, 100-yr [c]	1	3	25

[a] Kammen and Lew (2005)
[b] Smith (1994)
[c] IPCC (2012)

Table B.3: Conventional cooking fuel energy efficiency (all study sites)

Item	Figure	Units	Reference
Charcoal	25%	%	Anozie et al. (2007)
Electricity	90%	%	Anozie et al. (2007)
Liquid propane gas (LPG) and biogas	73%	%	Anozie et al. (2007)

General assumptions: sewered and non-sewered systems, Zambia

Table B.4: Transportation and distance assumptions for construction materials: Decentralized sewered, Zambia and non-sewered, Zambia[a]

Description	Distance	Unit	Trans-port type	Notes / assumptions
Material: cement, sand, stones				
Manufacturer to supplier (road)	50	km	Truck > 20 tonnes	Assume distance from Kafue to Lusaka, Zambia (Google Maps)
Supplier to project site (road)	20	km	Truck 10 – 20 tonnes	Assume project site within 20km of Lusaka City Center
Material: bricks				
Manufacturer to project site (road)	20	km	Truck 10 – 20 tonnes	Assume bricks sourced directly from manufacturer; within 20km of Lusaka City Center
Material: Metals and plastics (assume manufactured outside of Zambia)				
Manufacturer to supplier (sea)	15590	km	Container ship, ocean	Assume Shanghai, China to Durban, South Africa (overseas shipping: 8418 nautical miles) Source: http://ports.com/sea-route/port-of-shanghai,china/port-of-durban,south-africa/
Manufacturer to supplier (road)	2290	km	Truck >20 tonnes	Assume distance from Durban, South Africa to Lusaka, Zambia (Google Maps)
Supplier to project site (road)	20	km	Truck <10 tonnes	Assume project site within 20km of Lusaka City Center
Disposal of surplus soil				
Disposal of surplus soil	10	km	Truck 10– 20 tonnes	Assume surplus soil is disposed of within 10km of project site

[a]All data from BORDA (2016) unless noted otherwise

Table B.5: General population and infrastructure assumptions: Decentralized sewered, Zambia and non-sewered, Zambia

Item	Figure	Units	Reference
Population assumptions [a]			
COD generation *per capita*	61.9	gPCD	Laramee et al. (2017)
Nitrogen (as ammonia, NH4-N) generation *per capita*	6.3	gPCD	Laramee et al. (2017)
Mean weight *per* person (labourer / site worker)	60.7	kg	Walpole et al. (2012)
Infrastructure lifespan assumptions			
Main infrastructure lifespan (treatment infrastructure, sewerage)	20	years	BORDA – personal communication (2016); Sasse (1998)
On-site sanitation (pit latrines)	14	years	Average of various literature values: Hutton and Vargues (2016); WRC (2007); Tilley et al. (2014)
Biogas piping and stove infrastructure lifespan	6	years	Sasse (1998)
Equipment and tools (e.g. carts, barrels and tools for pit emptying)	6	years	Sasse (1998)
Electricity generation			
Electricity (hydroelectric facility, tropical reservoir)	96%	%	ZDA (2014)
Electricity (diesel or oil-fired power plant)	4%	%	ZDA (2014)
Human energy assumptions			
Human power - low intensity (e.g. cleaning)	3.5	MET	Ainsworth et al. (2016); Code: 05020
Human power - low intensity (e.g. walking)	4.5	MET	Ainsworth et al. (2016); Code: 17262
Human power - medium intensity (e.g. digging)	5	MET	Ainsworth et al. (2016); Code: 08050
Human power - high intensity (e.g. pushing wheelbarrow)	5.5	MET	Ainsworth et al. (2016); Code: 08255

[a] Assume equal COD and NH_4-N generation *per capita* for populations served by sewered and non-sewered systems for purposes of comparison

Table B.6: General financial assumptions: Decentralized sewered, Zambia and non-sewered, Zambia

Item	Figure	Units	Reference
Inflation, GDP deflator (Zambia: 2012; base year 2000)	503	-	World Bank (2015) [a]
Inflation, GDP deflator (Zambia: 2015; base year 2000)	618	-	World Bank (2015) [a]
Purchasing power parity (PPP) factor, GDP (LCU *per* international $) (Zambia: 2015)	2.9	-	World Bank (2015) [b]
Discount rate (local currency financial flows)	5%	%	World Bank (2013); Hutton and Varughese (2016)
Discount rate (USD currency financial flows)	1.2%	%	OMB (2015)
% cost for construction (labour, supervision, transport) - % of material list cost	50%	%	BORDA, personal communication (2016)
% cost for overall project management, planning and administration (% of total construction)	25%	%	BORDA, personal communication (2016)
% cost for technical planning and engineering design (% of total construction)	5%	%	BORDA, personal communication (2016)

[a] Databank code: Inflation, GDP deflator (annual %). From: http://data.worldbank.org/indicator/NY.GDP.DEFL.KD.ZG

[b] Databank code: Purchasing power parity (PPP). From: http://data.worldbank.org/indicator/PA.NUS.PPP

[c] Additional cost for project management and technical planning not included for cost of on-site sanitation infrastructure.

Decentralized sewered system, Zambia: LCA model inputs and findings

Table B.7: Construction materials and processes: Decentralized sewered, Zambia[a]

Item[b, c]	Quantity	Unit
[C1] - Household sewer line (total for 78 HH connections)		
Site clearance	281	m2
Excavate 200mm top soil	281	m2
Excavation up to 3m depth	56	m3
Ground leveling after excavation	281	m2
Backfilling and carefully compacting in layers	112	m3
Disposal of surplus soil	0	m3
PVC Pipe	1241	kg
[C2] - Main sewer line piping (total for 1488m piping)		
Site clearance	595	m2
Excavate 200mm top soil	595	m2
Excavation up to 3m depth	476	m3
Ground leveling after excavation	595	m2
Backfilling and carefully compacting in layers	595	m3
Disposal of surplus soil	0	m3
PVC Pipe	3794	kg
[C2] - Main sewer line manholes (total for 39 units)		
Site clearance	39	m2
Excavate 200mm top soil	8	m2
Excavation up to 3m depth	51	m3
Ground leveling after excavation	39	m2
Backfilling and carefully compacting in layers	59	m3
Disposal of surplus soil	0	m3
Cement	21342	kg
Sand	46966	kg
Aggregate	46037	kg
Reinforcing steel	337	kg
[C3] - Conveyance between secondary (ABR) and tertiary (PGF) treatment steps		
Site clearance	12	m2
Excavate 200mm top soil	12	m2
Excavation up to 3m depth	10	m3
Ground leveling after excavation	12	m2
Backfilling and carefully compacting in layers	12	m3
Disposal of surplus soil	0	m3
Cement	2736	kg
Sand	6021	kg
Aggregate	5902	kg
Reinforcing steel	43	kg
PVC Pipe	77	kg
[T1] - Biogas digester, dome construction (per unit) (Volume = 20m3) (2 Units)		
Site clearance	38	m2
Excavate 200mm top soil	38	m2
Excavation up to 3m depth	103	m3
Ground leveling after excavation	17	m2
Backfilling and carefully compacting in layers	21	m3
Disposal of surplus soil	81	m3
Cement	3246	kg
Sand	6288	kg
Aggregate	3134	kg

Bricks	9667	kg
Reinforcing steel	39	kg
Galvanized steel	3	kg
[T1] - Biogas digester, dome construction (per unit) (Volume = 30m3) (1 Unit)		
Site clearance	44	m2
Excavate 200mm top soil	44	m2
Excavation up to 3m depth	125	m3
Ground leveling after excavation	21	m2
Backfilling and carefully compacting in layers	28	m3
Disposal of surplus soil	96	m3
Cement	3746	kg
Sand	7088	kg
Aggregate	3534	kg
Bricks	11033	kg
Reinforcing steel	42	kg
Galvanized steel	3	kg
[T1] - Biogas digester, dome construction (per unit) (Volume = 40m3) (2 Units)		
Site clearance	50	m2
Excavate 200mm top soil	50	m2
Excavation up to 3m depth	149	m3
Ground leveling after excavation	24	m2
Backfilling and carefully compacting in layers	36	m3
Disposal of surplus soil	113	m3
Cement	4196	kg
Sand	8088	kg
Aggregate	4034	kg
Bricks	12524	kg
Reinforcing steel	44	kg
Galvanized steel	3	kg
[T2] - Anaerobic baffled reactor (per unit) (120m3) (1 Unit)		
Site clearance	247	m2
Excavate 200mm top soil	247	m2
Excavation up to 3m depth	621	m3
Ground leveling after excavation	222	m2
Backfilling and carefully compacting in layers	68	m3
Disposal of surplus soil	553	m3
Cement	47800	kg
Sand	144400	kg
Aggregate	87100	kg
Reinforcing steel	6626	kg
PVC Pipe	903	kg
[T3] - Horizontal planted gravel filter (per unit) (10m x 32m x 0.6m depth) (1 Units)		
Site clearance	320	m2
Excavate 200mm top soil	320	m2
Excavation up to 3m depth	256	m3
Ground leveling after excavation	320	m2
Backfilling and carefully compacting in layers	320	m3
Disposal of surplus soil	320	m3
Cement	6089	kg
Sand	63505	kg
Aggregate	323289	kg
Reinforcing steel	220	kg
PVC Pipe	62	kg
HDPE liner	686	kg
[D/R] - Biogas reuse infrastructure (total for 10 household connections)		

Site clearance	90	m2
Excavate 200mm top soil	90	m2
Excavation up to 3m depth	18	m3
Ground leveling after excavation	90	m2
Backfilling and carefully compacting in layers	18	m3
Disposal of surplus soil	0	m3
Galvanized steel piping	551	kg
Steel (cold rolled steel)	24	kg
Aluminum (stove burner head)	2	kg
Rubber hose pipe (PVC)	6	kg

[a] All data from BORDA (2016) unless otherwise specified
[b]Assume density of loose soil = 1200kg/m3
[c]Assume density of crushed stone and sand = 1600 kg/m3

Table B.8: Land area: Decentralized sewered, Zambia[a]

Item	**Land area (m2)**	**Number of units**	**Total land area (m2)**
[C1] - Household sewer line	N/A	-	-
[C2] - Main sewer line piping	N/A	-	-
[C3] – Piping and manholes (ABR to PGF)	N/A	-	-
[T1] - Biogas digester (*per* unit) (Volume = 20m3) (2 Units)	23	2	46
[T1] - Biogas digester (*per* unit) (Volume = 30m3) (1 Unit)	28	1	28
[T1] - Biogas digester (*per* unit) (Volume = 40m3) (2 Units)	32	2	65
[T2] - Anaerobic baffled reactor (*per* unit) (120m3) (1 Unit)	246	1	246
[T3] -Horizontal planted gravel filter (1 Unit)	384	1	384
[D/R] – Biogas piping & stoves	N/A	-	-
TOTAL LAND AREA			769

[a]All data from BORDA (2016) unless otherwise specified

Table B.9: System and site description: Decentralized sewered, Zambia

Item	**Figure**	**Units**	**Reference**
Year of project construction	2012	Year	BORDA (2016)
Number of people served	394	people	Laramee et al. (2017)
Number of households (wastewater connections)	78.8	households	Calculated
Number of people *per* household	5	people / HH	Laramee et al. (2017)
Wastewater generation per *capita*	74.4	LPCD	Laramee et al. (2017)
COD generation *per capita*	61.9	gPCD	Laramee et al. (2017)
Nitrogen generation *per capita*	6.3	gPCD	BORDA (2016)
Site elevation	1650	m	FloodMap.net (2014)
Site mean annual temperature	22.3	°C	World Bank (2012)
Water source: borehole (depth, m)	70	m	Assumed to be same as NS, ZMB system (no data available)
Water supply energy factor	0.40	kWh/m3	Calculated based on borehole depth; NWASCO (2016)
Pumping (wastewater) [a]	0	kWh/year	BORDA (2016)

[a] Gravity system for wastewater collection therefore no energy required.

Table B.10: Wastewater, chemical oxygen demand (COD) inputs and outputs *per* SSC functional group: Decentralized sewered, Zambia

Item	Figure	Units	Reference
Mean wastewater flow *per* day	29.3	m^3/day	Laramee et al. (2017)
Total biodegradable organic waste (TOW)	24.4	kg COD/day	Laramee et al. (2017)
Total organic nitrogen loading	2.5	kg N/day	BORDA (2016)
$COD_{(in,T1)}$	832	mg COD/L	Laramee et al. (2017)
$COD_{(removed,T1)}$	506	mg COD/L	Laramee et al. (2017)
$COD_{(out,T1)}$ / $COD_{(in,T2)}$	326	mg COD/L	Laramee et al. (2017)
$COD_{(removed,T2)}$	183	mg COD/L	BORDA (2016)
$COD_{(out,T2)}$ / $COD_{(in,T3)}$	144	mg COD/L	BORDA (2016)
$COD_{(removed,T3)}$	83	mg COD/L	BORDA (2016)
$COD_{(out,T3)\ (\text{*final discharge to stream})}$	61	mg COD/L	BORDA (2016)

Table B.11: CH_4 and N_2O production *per* functional group: Decentralized sewered, Zambia

Item	Figure	Units	Reference
[T1]: Biogas digesters			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: "Anaerobic reactor"
CH_4 production, gas phase (T1)	3.0	kg CH_4/day	Laramee et al. (2017)
CH_4 production, liquid phase (T1)	0.3	kg CH_4/day	Laramee et al. (2017)
Methane content	83%	%	Calculated
[T2]: Anaerobic baffled reactor			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: "Anaerobic reactor"
CH_4 production, gas phase (T2)	0.8	kg CH_4/day	Calculated
CH_4 production, liquid phase (T2)	0.4	kg CH_4/day	Calculated
Methane content	80%	%	Calculated
T3: Planted gravel filter (horizontal subsurface flow (HSSF)			
Methane correction factor (MCF)	0.1	-	IPCC (2013): Table 6.4: "Constructed wetland: HSSF"
CH_4 production, gas phase (T2)	0.03	kg CH_4/day	Calculated
CH_4 production, liquid phase (T2)	0.03	kg CH_4/day	Calculated
Methane content	70%	%	Calculated
Default emission factor (EF T-N_2O)	0.0079	kg N_2O/ kg N	IPCC (2013): Table 6.7 (Constructed wetland: horizontal sub-surface flow)
N_2O production *per* day	0.0063	kg N_2O/day	Calculated (see methods)
D/R: Disposal (final discharge)			
Methane correction factor (MCF)	0.1	-	IPCC (2006): Table 6.3: "Sea, river and lake discharge"
CH_4 production, gas phase (D/R)	0.01	kg CH_4/day	Calculated
CH_4 production, liquid phase (D/R)	0.04	kg CH_4/day	Calculated
Methane content	47%	%	Calculated
Default emission factor (EF T-N2O)	0.005	kg N_2O/ kg N	IPCC (2006): Table 6.11 (Wastewater discharged into aquatic environments)
N_2O production *per* day	0.0155	kg N_2O/day	Calculated (see methods)

Table B.12: Infrastructure and equipment replacement, recurrent materials, maintenance and transport: Decentralized sewered, Zambia

Item	Figure	Units	Reference
Biogas piping and stove replacement [a]			
Energy use (total)	22,500	MJ	LCA analysis (present study)
Emissions impact (total)	1860	$kgCO_2e$	LCA analysis (present study)
Recurrent materials [b]			
Cement: amount *per* year	50	kg/year	NWWSSC (2016); SimaPro: "Cement, Portland {RoW} | market for | Alloc Def, S"
Personal protection equipment (boots and gloves) - PVC: amount *per* year	10.4	kg/year	NWWSSC (2016); SimaPro: "Polyvinylchloride resin (S-PVC), suspension polymerisation, production mix, at plant, RER"
Work uniforms (coveralls) – cotton: amount *per* year	1.32	kg/year	NWWSSC (2016); SimaPro: Textile, "woven cotton {GLO}|market for | Alloc Def, S"
Tools (pipe wrench, shovel, etc.) – steel: amount *per* year	33	kg/year	NWWSSC (2016); SimaPro: Reinforcing steel {GLO}| market for | Alloc Def, S
Treatment infrastructure desludging (maintenance)			
T1: Biogas digester desludging: volume of sludge [c]	24	m3/ sludge year	BORDA (2016); SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S
T1: Biogas digester desludging: transport to disposal site [d]	600	tonne-km / year	BORDA (2016); NWWSSC (2016); SimaPro: SimaPro: Transport, truck <10t, EURO5, 80%LF, empty return
Other transport			
Vehicle travel (passenger vehicle, diesel) [e]	240	km / year	BORDA (2016)

[a] See general assumptions for assumed lifespan. Cost *per capita per* year for infrastructure or equipment replacement is calculated as: Total impact (energy or emissions*number of replacement events)/(number of people served * analysis period).
[b] Recurrent material impacts calculated as: (material amount *per* year * energy or emission factor) / number of people served.
[c] Assume complete removal of sludge from T1 (2*20m3 and 2*40m3 digesters) occurs 1 time *per* 5 years.
[d] Assume: 12km roundtrip to disposal site, 25trips required to desludge all digesters, 10 tonne load *per* trip (6tonne sludge + 4tonne truck) and one desludging event every 5 years.
[e] Assume 10km roundtrip distance to site and 2 trips *per* month = 240km *per* year.

Table B.13: Conventional cooking fuels: Decentralized sewered, Zambia

Item	Figure	Units	Reference
Conventional fuel used for cooking energy			
Charcoal	78%	%	Laramee et al. (2017)
Electricity	22%	%	Laramee et al. (2017)
Energy content of cooking fuels			
Charcoal: energy content	25.9	MJ/kg charcoal (dry weight)	Laramee et al. (2017)
Electricity: energy content	3.6	MJ/kWh	Standard definition
Methane: energy content	55.5	MJ/kg	Standard definition

Table B.14: Capital expenditure (CapEx): Decentralized sewered, Zambia[a]

Item	Figure	Units	Reference
User CapEx			
[C1] - Household sewer line (total for 78 HH connections)	76,008	ZMW	BORDA (2016)
Agency CapEx			
[C2] - Main sewer line piping (1488m piping) & manholes (39 units)	408,241	ZMW	BORDA (2016)
[C3] – Piping and manholes at ABR and PGF	22,839	ZMW	BORDA (2016)
[T1] - Biogas digesters (total for (2) 20m3 volume, (2) 40m3 volume, (1) 30m3 volume)	263,891	ZMW	BORDA (2016)
[T2] - Anaerobic baffled reactor (*per* unit) (120m3) (1 Unit)	604,082	ZMW	BORDA (2016)
[T3] -Horizontal planted gravel filter (*per* unit) (10m x 32m x 0.6m depth) (1 Units)	335,176	ZMW	BORDA (2016)
[D/R] – Biogas piping and stoves (total for 10 household connections)	39,238	ZMW	BORDA (2016)

[a] All costs adjusted to Year 2015 ZMW using GDP deflator and include estimated costs for planning and technical design.

Table B.15: Operational expenditures (OpEx): Decentralized sewered, Zambia[a]

Item	Figure	Units	Reference
User OpEx - costs			
Water supply tariff (wastewater only) [b]	(112)	ZMW/cap/yr	Calculated based on Laramee et al. (2017) and NWASCO (2015)
Sanitation surcharge tariff [c]	(64)	ZMW/cap/yr	Calculated based on Laramee et al. (2017) and NWASCO (2015)
Fixed charge	(10)	ZMW/cap/yr	NWASCO (2015)
Agency OpEx - costs			
Human resource costs	(83)	ZMW/cap/yr	NWWSSC (2016)
Material costs	(6)	ZMW/cap/yr	NWWSSC (2016)
Energy and fuel costs	(1)	ZMW/cap/yr	NWWSSC (2016)
Water supply cost (wastewater only) [b, d]	(112)	ZMW/cap/yr	NWWSSC (2016)
Agency OpEx - benefits			
Water supply tariff (wastewater only) [b, d]	112	ZMW/cap/yr	NWWSSC (2016)
Sanitation surcharge tariff	64	ZMW/cap/yr	NWWSSC (2016)
Fixed charge	10	ZMW/cap/yr	NWWSSC (2016)
Biogas tariff (theoretical)	0.15	ZMW/MJ biogas	Laramee et al., 2017

[a] All costs and benefits are reported in Year 2015 ZMW (adjusted using GDP deflator)
[b] Cost of water supply calculated using mean wastewater generated *per capita per* day and the second block tariff in the year 2015 (NWASCO, 2015)
[c] Sanitation surcharge is calculated as 40% of total water supply charge. Water supply is estimated assuming wastewater generation is 50-75% water supplied to household (UNEP/WHO, 1997). All increasing block tariffs are then used to calculate the mean household water bill (NWASCO, 2015).
[d] Agency cost to supply water assumed to be equal to benefit generated from water tariff
[e] Theoretical biogas tariff calculated *per* energy content, fuel efficiency and cost of conventional fuels (Laramee et al., 2017)

Table B.16: Capital maintenance expenditures (CapManEx): Decentralized sewered, Zambia[a]

Item	Figure	Units	Reference
User CapManEx			
None	-	-	-
Agency CapManEx			
Desludging digesters	(34)	ZMW/cap *per* 5yrs	BORDA (2016)
PGF filter cleaning	(53)	ZMW/cap *per* 5yrs	BORDA (2016)
Gas stove replacement	(20)	ZMW/cap *per* 6yrs	BORDA (2016)
Gas piping replacement	(61)	ZMW/cap *per* 6yrs	BORDA (2016)

[a] All costs and benefits are reported in Year 2015 ZMW (adjusted using GDP deflator)

Table B.17: Energy use findings by functional group: Infrastructure, use phase and net impacts: Decentralized sewered, Zambia

Item	**Containment (MJ/*capita*/ year)**	**Conveyance (MJ/*capita*/ year)**	**Treatment (MJ/*capita*/ year)**	**Disposal / Reuse (MJ/*capita*/ year)**	**Net (MJ/*capita*/ year)**
Infrastructure	0.0	76.9	144.5	2.9	224.3
Use phase	0.0	38.9	18.3	-303.6	-246.4
Net (mean)	0.0	115.7	162.9	-300.8	-22.1
Lower bound uncertainty	0.0	100.1	137.2	-587.2	-349.9
Upper bound uncertainty	0.0	142.1	209.5	-148.0	203.6

Table B.18: Global warming potential (GWP) findings by functional group: Infrastructure, use phase and net impacts: Decentralized sewered, Zambia

Item	**Containment ($kgCO_2e$/ *capita*/year)**	**Conveyance ($kgCO_2e$/ *capita*/year)**	**Treatment ($kgCO_2e$/ *capita*/year)**	**Disposal / Reuse ($kgCO_2e$/ *capita*/year)**	**Net ($kgCO_2e$/ *capita*/year)**
Infrastructure	0.0	7.0	19.8	0.2	27.0
Use phase	0.0	18.7	52.8	-76.8	-5.4
Net (mean)	0.0	25.6	72.6	-76.6	21.6
Lower bound uncertainty	0.0	23.0	71.1	-148.6	-54.5
Upper bound uncertainty	0.0	29.2	75.0	-34.6	69.6

Table B.19: Infrastructure phase energy use and global warming potential (GWP) by source of impact: Decentralized sewered system, Zambia

Item	**Energy use (MJ/*capita*/ year)**	**Percentage of energy use [a]**	**GWP ($kgCO_2e$/ *capita*/year)**	**Percentage of GWP (%) [a]**
Cement (I)	70.4	24%	16.9	10%
Steel (I)	27.7	10%	2.4	2%
Clay bricks (I)	18.0	6%	1.8	1%
PVC Pipe / plastics (I)	59.1	20%	2.4	1%
Transportation (I)	30.9	11%	2.2	1%
Construction works (I)	17.9	6%	1.3	1%
Other (I)	0.1	0%	0.1	0%

[a] Percentage of lifecycle energy use or GWP impacts excluding reductions in energy use or GWP due to recovered CH_4 or co-product offsets

Table B.20: Use phase energy by source of impact: Decentralized sewered, Zambia

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use (%) [a]**
EE,j - Electricity use (U)	0.0	0%
EM,j - Material use (U)	14.1	5%
EW,j - Water supply (U)	38.9	13%
EV,j - Vehicle use (U)	12.8	4%
EH,j - Human energy (U)	0.0	0%
EI,j - Co-products (U)	-312.2	-

[a] Percentage of lifecycle energy use excluding reductions due to co-product offsets

Table B.21: Use phase global warming potential (GWP) by source of impact: Decentralized sewered, Zambia

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use [a]**
CE,j - Electricity inputs (U)	0.0	0%
CM,j - Material inputs (U)	1.2	1%
CW,j - Water supply (U)	18.7	12%
CV,j - Vehicle use (U)	1.0	1%
CP-CH_4,j - Treatment (CH_4) (U)	107.9	67%
CP-N_2O,j - Treatment (N_2O) (U)	6.0	4%
CR,j - Recovered CH_4 (U)	-57.3	-
CC,j - Co-products (U)	-82.9	-

[a] Percentage of lifecycle GWP excluding reductions due to recovered CH_4 or co-product offsets

Table B.22: User equivalent annual cost (EAC): Decentralized sewered, Zambia

Item	**User EAC - mean (USD/*capita*/year) [a]**	**User EAC - lower bound (USD/*capita*/year) [a]**	**User EAC - upper bound (USD/*capita*/year) [a]**
CapEx	$(5)	$(8)	$(4)
OpEx - Costs	$(63)	$(83)	$(28)
OpEx - Benefits	$-	$-	$-
CapManEx	$-	$-	$-
Carbon financing	$-	$-	$-
Net User EAC	$(68)	$(91)	$(32)

[a] All costs in Year 2015 USD

Table B.23: Agency equivalent annual cost (EAC): Decentralized sewered, Zambia

Item	**Agency EAC - mean (USD/*capita*/year) [a]**	**Agency EAC - lower bound (USD/*capita*/year) [a]**	**Agency EAC - upper bound (USD/*capita*/year) [a]**
CapEx	$(116)	$(206)	$(71)
OpEx - Costs	$(69)	$(70)	$(62)
OpEx - Benefits	$69	$44	$71
CapManEx	$(8)	$(9)	$(7)
Carbon financing	$-	$-	$-
Net Agency EAC	$(124)	$(241)	$(69)

[a] All costs in Year 2015 USD

Non-sewered system, Zambia: LCA model inputs and findings

Table B.24: Construction materials and processes: Non-sewered, Zambia[a]

Item[b, c]	Quantity	Unit
Pit latrine (1 concrete block latrine)		
Site clearance	1.2	m2
Excavate 200mm top soil	0.2	m2
Excavation up to 3m depth	2.4	m3
Ground leveling after excavation	1.2	m2
Backfilling and carefully compacting in layers	0.8	m3
Disposal of surplus soil	2.4	m3
Cement	251	kg
Sand	546	kg
Aggregate	1197	kg
Reinforcing steel	8.9	kg
Primary conveyance: (steel cart (4 No.), steel emptying tools, barrels (60 No.)		
Steel (reinforcing steel)	194	kg
Synthetic rubber	9	kg
Polyethylene	31	kg
Steel (chromium)	25	kg
HDPE	150	kg
Secondary conveyance (vacuum tanker)		
16ton lorry assumed in SimaPro (1/2 day *per* week)		
Biogas digester: dome construction and expansion chamber (per unit) (Volume = 56m3) (1 Units)		
Site clearance	89	m2
Excavate 200mm top soil	89	m2
Excavation up to 3m depth	225	m3
Ground leveling after excavation	59	m2
Backfilling and carefully compacting in layers	84	m3
Disposal of surplus soil	174	m3
Cement	9563	kg
Sand	18458	kg
Aggregate	18047	kg
Bricks	14945	kg
Reinforcing steel	196	kg
Galvanized steel	3	kg
Site infrastructure (ramp, road, solid waste drying racks)		
Site clearance	182	m2
Excavate 200mm top soil	182	m2
Excavation up to 3m depth	7	m3
Ground leveling after excavation	182	m2
Backfilling and carefully compacting in layers	29	m3
Disposal of surplus soil	0	m3
Cement	12826	kg
Sand	23414	kg
Aggregate	32888	kg
Reinforcing steel	1194	kg
Polycarbonate roofing	64	kg
Sludge drying beds (35m2) per unit (12 units total)		
Site clearance	48	m2
Excavate 200mm top soil	35	m2
Excavation up to 3m depth	5	m3
Ground leveling after excavation	35	m2

Backfilling and carefully compacting in layers	5	m3
Disposal of surplus soil	0	m3
Cement	2293	kg
Sand	9499	kg
Aggregate	5018	kg
Reinforcing steel	5040	kg
PVC Pipe	597	kg
Biogas reuse infrastructure (1 institutional connection)		
Site clearance	30	m2
Excavate 200mm top soil	30	m2
Excavation up to 3m depth	6	m3
Ground leveling after excavation	30	m2
Backfilling and carefully compacting in layers	40	m3
Disposal of surplus soil	0	m3
Galvanized steel piping	170	kg
Steel (cold rolled steel)	24	kg
Aluminum (stove burner head)	2	kg
Rubber hose pipe (PVC)	6	kg

[a] All data from BORDA (2016) unless otherwise specified
[b] Assume density of loose soil = 1200kg/m3
[c] Assume density of crushed stone and sand = 1600 kg/m3

Table B.25: Land area: Non-sewered, Zambia[a]

Item	**Land area (m2)**	**Number of units**	**Total land area (m2)**
[C/S] - Pit latrine (concrete block)	N/A	-	-
[C1] – Steel carts	N/A	-	-
[C2] – Vacuum tanker	N/A	-	-
[T1] - Biogas digester (*per* unit) (Volume = 56m3) (1 Units)	107	1	107
[T1] – Site infrastructure (ramp, drying racks, etc.)	704	1	704
[T2] – Sludge drying beds (35m2) (12 Units)	58	12	691
[D/R] – Biogas piping & stoves (1 institutional connection)	N/A	-	-
TOTAL LAND AREA			1502

[a] All data from BORDA (2016) unless otherwise specified

Table B.26: System and site description: Non-sewered, Zambia

Item	Figure	Units	Reference
Year of project construction	2012	Year	WASAZA/BORDA (2013)
Mean TS of fecal sludge	17%	%	WASAZA/BORDA (2013)
Mean kg fecal sludge input *per* day (wet wt)	912	kg/day	WASAZA/BORDA (2013)
Mean kg fecal sludge input *per* day (dry wt)	150	kg/day	Calculated
Total organic solids produced *per capita per* day	39	g/cap/day	Rose et al. (2016)
Depth of pit emptied	1m	m	WASAZA/BORDA (2013)
Number of people served	12432	people	Calculated
Mean number of people sharing toilet [a]	5	people / HH	Laramee et al. (2017)
Number of toilets	2486	toilets	Calculated
Wastewater generation *per capita*	0	LPCD	WASAZA/BORDA (2013)
COD generation *per capita* [b]	62	gPCD	Laramee et al. (2017)
Nitrogen generation *per capita*	6.3	gPCD	BORDA (2016)
Site elevation	1280	m	FloodMap.net (2014)
Site mean annual temperature	22.3	°C	World Bank (2012)
Borehole water source (depth, m)	70	m	KWT (2016)
Energy req'd for water supply (borehole)	0.40	kWh/m3	Calculated based on borehole depth and assumed pump efficiencies

[a] Measured mean number of people *per* latrine: 17.7 (SD=8.6) (WSUP, 2016). However, mean number of people *per* household from DS, Zambia used in baseline analysis for standard comparison across systems and to meet SDG criteria for improved sanitation (not shared).
[b] *Per-capita* COD assumed to be equivalent to COD *per capita* measured for sewered system (no data available specific to site)

Table B.27: Chemical oxygen demand (COD) inputs and outputs *per* SSC functional group: Non-sewered, Zambia

Item	Figure	Units	Reference
Total organic waste (TOW), COD(in,C/S)	771	kg COD/day	Calculated (see methods)
Total organic nitrogen loading	78	kg N / day	Calculated
$COD_{(removed,C/S)}$	680	kg COD / day	Calculated [a]
$COD_{(in,T1)}$	90	kg COD / day	Calculated
$COD_{(removed,T1)}$	20	kg COD / day	Calculated based on measured biogas production at T1
$COD_{(in,T2)}$	71	kg COD / day	Calculated
$COD_{(removed,T2)}$	62	kg COD / day	Calculated based on Koottatep et al. (2005)
$COD_{(in, D/R)}$ (final discharge to land application)	8	kg COD / day	Calculated
$COD_{(removed,D/R)}$	8	kg COD / day	Calculated (all COD assumed to be removed)

[a] Calculated based on mean and range of kg fecal sludge inputs to treatment plant *per* day, total solids content (%) and dry weight of organic solids produced *per capita per* day (see methods).

Table B.28: CH_4 and N_2O production *per* functional group: Non-sewered, Zambia

Item	Figure	Units	Reference
[C/S]: Household pit latrine (or septic tank)			
Methane correction factor (MCF)	0.5	-	IPCC (2006): Chapter 6, Table 6.3: "latrine - dry, >5 users"
Biogas production (gas phase)	85.3	kg CH_4/day	Calculated (see methods)
Methane content	71%	%	Calculated (see methods)
[T1]: Biogas digester			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: "Anaerobic reactor"
CH4 production, gas phase (T1)	4.5	kg CH_4/day	WASAZA/BORDA (2013)
Methane content	71%	%	Calculated (see methods)
[T2-S]: Sludge drying beds			
Methane correction factor (MCF)	0.2	-	IPCC (2006): Table 6.3: "Anaerobic shallow lagoon"
Biogas production (gas phase)	3.1	kg CH_4/day	Calculated (see methods)
Methane content	71%	%	Calculated (see methods)
Default emission factor (EF T-N_2O)	0.005	kg N_2O/ kg N	IPCC (2006): Chapter 10, Table 10.21 (Solid storage)
N_2O production *per* day	0.392	kg N_2O/day	Calculated (see methods)
D/R: Disposal (final discharge)			
Methane correction factor (MCF)	0.01	-	IPCC (2006): Chapter 10, Table 10.17: "composting: passive winrow"
CH4 production, gas phase (D/R)	0.02	kg CH_4/day	Calculated (see methods)
Methane content	71%	%	Calculated (see methods)

[a] Assume methane production in liquid phase negligible COD concentration > 5000mg/L

Table B.29: Conveyance: Water supply, motorized and manual conveyance inputs: Non-sewered, Zambia

Item	Figure	Units	Reference
[C/S]: Household pit latrine (or septic tank) inputs			
Water supply: borehole [a]	0	m3/day	KWT (2016)
C1: Primary Conveyance - Inputs			
Water inputs: borehole [b]	1.4	m3/day	KWT (2016)
Pumping/desludging (on-site sanitation) [c]	0	m3 sludge/ year	KWT (2016)
Vehicle – light duty (<10tonne) [d]	0	tonne-km/year	KWT (2016)
Vehicle – heavy duty (10-20tonne) [d]	0	tonne-km/year	KWT (2016)
Human power - low intensity (e.g. cleaning) e	2080	Person-hrs/year	KWT (2016)
Human power - low intensity (e.g. walking) f	2080	Person-hrs/year	KWT (2016)
Human power - medium intensity (e.g. digging) g	4160	Person-hrs/year	KWT (2016)
Human power - high intensity (e.g. pushing wheelbarrow) h	2080	Person-hrs/year	KWT (2016)
C1: Secondary Conveyance - Inputs			
Water inputs: borehole	0	m3/day	KWT (2016)
Vehicle - light duty (<10t tonne)	0	tonne-km/year	WASAZA/BORDA (2013); SimaPro: Transport, truck < 10t, EURO5, 80%LF, empty return
Vehicle - heavy duty (10-20t tonne) [i]	8008	tonne-km/year	WASAZA/BORDA (2013) ; SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"
Pumping/desludging [j]	364	m3 sludge/ year	WASAZA/BORDA (2013); SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S

[a] Assume no water used for sanitation (household toilets typically dry pit latrines

[b] Kanyama pit emptying team report adding 1 barrel water for every 6 barrels of fecal sludge emptied (to liquefy pit contents to aid with emptying) and 1 barrel of water for washing for every 6 barrels emptied at treatment system. Addlitional 1m3 used *per* day at treatment site for cleaning equipment, etc.

[c] Desludging of on-site pit latrines is performed manually (no mechanical energy required)

[d] Primary conveyance is performed manually (no vehicular energy required)

[e] Assumption: 1hr/day washing*4 people*2 sites/day*5days/week*52 weeks/year

[f] Assumption: 1hr/day walking to site (no load)*4 people*2 sites/day*5days/week*52 weeks/year

[g] Assumption: 2hr/day pit-emptying*4 people*2 sites/day*5days/week*52 weeks/year

[h] Assumption: 1hr/day walking back to site (with load)*4 people*2 sites/day*5days/week*52 weeks/year

[i] Vehicle load (C2) based on 14km/roundtrip from primary to secondary treatment facility; 1 trip *per* week*52 weeks *per* year*(7000kg load + 4000kg vehicle weight)

[j] Pumping / desludging (C2): based on one desludging event and one discharge event *per* week of 7000L sludge (52 weeks *per* year)

Table B.30: Infrastructure and equipment replacement, recurrent materials, maintenance and transport: Non-sewered, Zambia

Item	Figure	Units	Reference
On-site sanitation (pit latrine) replacement [a]			
Energy use *per* latrine	2090	MJ/unit	Calculated [e]
Emissions *per* latrine	401	$kgCO_2e$/unit	Calculated [e]
Conveyance equipment (cart and tools) replacement (C1) [a]			
Energy use (total for equipment)	25,400	MJ / replacement	Calculated [e]
Emissions impact (total for equipment)	1,160	$kgCO_2e$ / replacement	Calculated [e]
Biogas piping and stove replacement [a]			
Energy use (total)	8,340	MJ / replacement	Calculated [e]
Emissions impact (total)	674	$kgCO_2e$ / replacement	Calculated [e]
Recurrent materials [b]			
Cement: amount *per* year	2,600	kg/year	WASAZA/BORDA (2013) SimaPro: "Cement, Portland {RoW} | market for | Alloc Def, S"
Cleaning disinfectant: amount *per* year	104	kg/year	WASAZA/BORDA (2013) SimaPro: Sodium hypochlorite, without water, in 15% solution state {GLO}|market for | Alloc Def, S
Personal protection equipment (boots and gloves) -PVC: amount *per* year	52	kg/year	WASAZA/BORDA (2013) SimaPro: Polyvinylchloride resin (S-PVC), suspension polymerisation, production mix, at plant, RER
Work uniforms (coveralls) – cotton: amount *per* year	26.4	kg/year	WASAZA/BORDA (2013) SimaPro: Textile, "woven cotton {GLO}|market for | Alloc Def, S"
Treatment infrastructure desludging (maintenance)			
T1: Biogas digester desludging [c]	112	m3/ sludge year	WASAZA/BORDA (2013) SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S
T1: Biogas digester desludging: transport to disposal site [d]	2464	tonne-km / year	WASAZA/BORDA (2013) Transport, truck < 10t, EURO5, 80%LF, empty return

[a] See general assumptions for assumed lifespan. Cost *per capita per* year for infrastructure or equipment replacement is calculated as: Total impact (energy or emissions*number of replacement events)/(number of people served * analysis period).
[b] Recurrent material impacts calculated as: (material amount *per* year * energy or emission factor) / number of people served.
[c] Assume complete removal of sludge from T1 (biogas digester, volume = 56m3) occurs twice *per* year.
[d] Assume: 14km roundtrip to disposal site, 8 trips required to desludge digester, 11 tonne load *per* trip (7 tonne sludge + 4tonne truck) and two desludging events every year.
[e] Calculated in SimaPro Version 8.3. See previous "Construction materials and processes" table for more information on material use.

Table B.31: Conventional cooking fuels: Non-sewered, Zambia

Item	Figure	Units	Reference
Conventional fuel used for cooking energy			
Charcoal	100%	%	WASAZA/BORDA (2013)
Energy content of cooking fuels			
Charcoal: energy content	25.9	MJ/kg charcoal (dry weight)	Laramee et al. (2017)
Methane: energy content	55.5	MJ/kg	Standard definition

Table B.32: Capital expenditure (CapEx): Non-sewered, Zambia[a]

Item	Figure	Units	Reference
User CapEx			
[C/S] - Pit latrine (concrete block substructure) (*per* unit) [b]	1,866	ZMW	BORDA (2016)
Agency CapEx			
[C1] – Steel carts and emptying tools	45,993	ZMW	BORDA (2016)
[C2] – Vacuum tanker [c]	N/A	ZMW	BORDA (2016)
[T1] - Biogas digester (Volume = 56m3) (1 Unit) and site infrastructure (ramp, drying racks, etc.)	218,893	ZMW	BORDA (2016)
[T2] – Sludge drying beds (35m2) (*per* unit) (12 Units)	614,378	ZMW	BORDA (2016)
[D/R] – Biogas piping & stoves (1 institutional connection)	18,948	ZMW	BORDA (2016)

[a] All costs adjusted to Year 2015 ZMW using GDP deflator and include estimated costs for planning and technical design

[b] *Per-capita* cost of pit latrine calculated as cost *per* unit/mean number of people sharing latrine

[c] Vacuum tanker not owned by project. Payment to private contractor for each use; costs reflected in OpEx.

Table B.33: Operational expenditures (OpEx): Non-sewered, Zambia[a]

Item	Figure	Units	Reference
User OpEx - costs			
Water supply tariff (wastewater only)	N/A	-	None - assume dry pit latrines
Pit emptying fee [b]	14.9	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Agency OpEx - costs			
Human resource costs	19.8	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Material costs	4.8	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Energy and fuel costs	2.3	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Water supply cost	0.1	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Agency OpEx - benefits			
Pit emptying revenue [b]	15	ZMW/cap/yr	Calculated w/ data: WSUP (2016)
Biogas tariff (theoretical) [c]	0.18	ZMW/MJ	Laramee et al. (2017)

[a] All costs and benefits are reported in Year 2015 ZMW (adjusted using GDP deflator)
[b] Pit emptying fee is assumed as a 'cost' to the user and a 'benefit' to the agency. Figure is based on emptying charge of: 250 ZMW for 12 60L barrels, 380 ZMW for 24 60L barrels and 450 ZMW for 32 60L barrels emptied from pit (year 2013-14). Total revenue generated in 2013 is divided by the total number of users to calculate an average annual cost *per* user.
[c] Theoretical biogas tariff calculated *per* energy content, fuel efficiency and cost of conventional fuels (Laramee et al., 2017)

Table B.34: Capital maintenance expenditure (CapManEx): Non-sewered, Zambia[a]

Item	Figure	Units	Reference
User CapManEx			
Pit latrine replacement (*per* unit) [b]	1866	ZMW / unit	BORDA (2016)
Agency CapManEx			
Desludging digesters (agency)	1.0	ZMW, 2x *per* year	BORDA (2016)
Replace gas stoves and piping (agency)	1.5	ZMW / 6 years	BORDA (2016)
Replace carts and tools (agency)	3.7	ZMW / 6 years	BORDA (2016)

[a] All costs and benefits are reported in Year 2015 ZMW (adjusted using GDP deflator)
[b] An equal number of pit latrines are assumed to be replaced each year for the duration of the analysis period. Cost of pit latrine categorized as CapEx for assumed lifespan of pit and thereafter categorized as CapManEx. Annual *per-capita* cost for pit latrine replacement is calculated as: (unit cost of pit latrine) / (lifespan * mean number of people sharing pit latrine).

Table B.35: Energy use findings by functional group: Infrastructure, use phase and net impacts: Non-sewered, Zambia

Item	**Containment (MJ/*capita*/ year)**	**Conveyance (MJ/*capita*/ year)**	**Treatment (MJ/*capita*/ year)**	**Disposal / Reuse (MJ/*capita*/ year)**	**Net (MJ/*capita*/ year)**
Infrastructure	20.9	0.2	3.7	0.2	25.0
Use phase	9.0	7.3	1.5	-17.4	0.3
Net (mean)	29.9	7.5	5.1	-17.3	25.2
Lower bound uncertainty	24.0	2.6	2.4	-33.0	-4.1
Upper bound uncertainty	41.5	16.8	11.4	-8.1	61.6

Table B.36: Global warming potential (GWP) findings by functional group: Infrastructure, use phase and net impacts: Non-sewered, Zambia

Item	**Containment ($kgCO_2e$/ *capita*/year)**	**Conveyance ($kgCO_2e$/ *capita*/year)**	**Treatment ($kgCO_2e$/ *capita*/year)**	**Disposal / Reuse ($kgCO_2e$/ *capita*/year)**	**Net ($kgCO_2e$/ *capita*/year)**
Infrastructure	4.0	0.0	0.5	0.0	4.5
Use phase	64.3	0.6	6.4	-4.3	67.0
Net (mean)	68.3	0.6	6.9	-4.3	71.5
Lower bound uncertainty	32.9	0.6	7.4	-5.0	35.9
Upper bound uncertainty	111.2	0.6	6.4	-3.5	114.7

Table B.37: Infrastructure phase energy use and global warming potential (GWP) by source of impact: Non-sewered, Zambia

Item	**Energy use (MJ/*capita*/ year)**	**Percentage of energy use [a]**	**GWP (kgCO_2e/ *capita*/year)**	**Percentage of GWP (%) [a]**
Cement (I)	15.3	36%	3.8	5%
Steel (I)	3.3	8%	0.3	0%
Clay bricks (I)	0.8	2%	0.1	0%
PVC Pipe / plastics (I)	0.3	1%	0.0	0%
Transportation (I)	3.7	9%	0.2	0%
Construction works (I)	1.3	3%	0.1	0%
Other (I)	0.2	0%	0.0	0%

[a] Percentage of lifecycle energy use or GWP impacts excluding reductions in energy use or GWP due to recovered CH_4 or co-product offsets

Table B.38: Use phase energy use by source of impact: Non-sewered, Zambia

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use (%) [a]**
EE,j - Electricity use (U)	0.0	0%
EM,j - Material use (U)	11.9	28%
EW,j - Water supply (U)	0.1	0%
EV,j - Vehicle use (U)	4.9	11%
EH,j - Human energy (U)	1.0	2%
EI,j - Co-products (U)	-17.5	-

[a] Percentage of lifecycle energy use excluding reductions due to co-product offsets

Table B.39: Use phase global warming potential (GWP) by source of impact: Non-sewered, Zambia

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use [a]**
CE,j - Electricity inputs (U)	0.0	0%
CM,j - Material inputs (U)	2.1	3%
CW,j - Water supply (U)	0.0	0%
CV,j - Vehicle use (U)	0.3	0%
CP-CH_4,j - Treatment (CH_4) (U)	68.2	87%
CP-N_2O,j - Treatment (N_2O) (U)	3.4	4%
CR,j - Recovered CH_4 (U)	-2.7	0%
CC,j - Co-products (U)	-4.4	-

[a] Percentage of lifecycle GWP excluding reductions due to recovered CH_4 or co-product offsets

Table B.40: User equivalent annual cost (EAC): Non-sewered, Zambia

Item	**User EAC - mean (USD/*capita*/year)** [a]	**User EAC - lower bound (USD/*capita*/year)** [a]	**User EAC - upper bound (USD/*capita*/year)** [a]
CapEx	$(8)	$(11)	$(6)
OpEx - Costs	$(5)	$(14)	$(2)
OpEx - Benefits	$-	$-	$-
CapManEx	$(2)	$(5)	$(1)
Carbon financing	$-	$-	$-
Net User EAC	$(15)	$(30)	$(9)

[a] All costs in Year 2015 USD

Table B.41: Agency equivalent annual cost (EAC): Non-sewered, Zambia

Item	**Agency EAC - mean (USD/*capita*/year)** [a]	**Agency EAC - lower bound (USD/*capita*/year)** [a]	**Agency EAC - upper bound (USD/*capita*/year)** [a]
CapEx	$(2)	$(7)	$(1)
OpEx - Costs	$(9)	$(27)	$(3)
OpEx - Benefits	$5	$11	$3
CapManEx	$(2)	$(5)	$(1)
Carbon financing	$-	$-	$-
Net Agency EAC	$(7)	$(29)	$(1)

[a] All costs in Year 2015 USD

General assumptions: sewered and non-sewered systems, India

Table B.42: Transportation and distance assumptions for construction materials: Decentralized sewered, India and non-sewered, India

Description	**Distance**	**Unit**	**Trans-port type**	**Notes / assumptions**
Material: cement				
Manufacturer to supplier (road)	20	km	Truck > 20 tonnes	Assume manufactured in Bangalore (CDD, 2016)
Supplier to project site (road)	20	km	Truck 10–20 tonnes	Assume project site within 20km of Bangalore (CDD, 2016)
Material: stones, sand				
Manufacturer to supplier (road)	50	km	Truck > 20 tonnes	Assume manufactured near Bangalore
Supplier to project site (road)	20	km	Truck 10–20 tonnes	Assume project site within 20km of Bangalore (CDD, 2016)
Material: Metals and plastics				
Manufacturer to supplier (road)	10	km	Truck > 20 tonnes	Assume manufactured in Bangalore
Supplier to project site (road)	20	km	Truck <10 tonnes	Assume project site within 20km of Bangalore (CDD, 2016)
Disposal of surplus soil				
Disposal of surplus soil	10	km	Truck 10–20 tonnes	Assume surplus soil is disposed of within 10km of project site (CDD, 2016)

[a]*All data from CDD (2016) unless noted otherwise*

Table B.43: General population and infrastructure assumptions: Decentralized sewered, India and non-sewered, India

Item	**Figure**	**Units**	**Reference**
Population assumptions [a]			
COD generation *per-capita*	53.7	gPCD	Calculated using data from Reynaud and Buckley (2015) and Miller (2011)
Nitrogen generation *per-capita*	5.0	gPCD	Reynaud and Buckley, 2015
Mean weight *per* person (labourer / site worker)	57.7	kg / person	Walpole et al. (2012)
Infrastructure lifespan assumptions			
Main infrastructure lifespan (treatment infrastructure, sewerage)	20	years	BORDA – personal communication (2016); Sasse (1998)
On-site sanitation (pit latrines)	14	years	Average of various literature values: Hutton and Vargues (2016): 8yrs; WRC (2007): 10yrs; Tilley et al. (2014): 20yrs.
Biogas piping and stove infrastructure lifespan	6	years	Sasse (1998)
Equipment and tools (e.g. carts, barrels and tools for pit emptying)	6	years	Sasse (1998)
Electricity generation			

Electricity (hydroelectric facility, tropical reservoir)	15%	%	OECD/IEA (2015)
Electricity (coal-fired power plant)	60%	%	OECD/IEA (2015)
Electricity (diesel or oil-fired power plant)	3%	%	OECD/IEA (2015)
Electricity (natural gas fired power plant)	8%	%	OECD/IEA (2015)
Electricity (photovoltaic)	1%	%	OECD/IEA (2015)
Electricity (wind)	8%	%	OECD/IEA (2015)
Human energy assumptions			
Human power - low intensity (e.g. cleaning)	3.5	MET	Ainsworth et al. (2016); Code: 05020
Human power - low intensity (e.g. walking)		4.5	MET
Human power - medium intensity (e.g. digging)	5	MET	Ainsworth et al. (2016); Code: 08050
Human power - high intensity (e.g. pushing wheelbarrow)	5.5	MET	Ainsworth et al. (2016); Code: 08255

[a] Assume equal COD and NH_4-N generation *per capita* for populations served by sewered and non-sewered systems for purposes of comparison

Table B.44: General financial assumptions: Decentralized sewered, India and non-sewered, India

Item	Figure	Units	Reference
Inflation, GDP deflator (India: 2004; base year 2000)	117.6	-	World Bank (2015) [a]
Inflation, GDP deflator (India: 2015; base year 2000)	220.3	-	World Bank (2015) [a]
Purchasing power parity (PPP) factor, GDP (LCU *per* international $) (India: 2015)	17.0	-	World Bank (2015) [b]
Discount rate (local currency financial flows)	5%	%	World Bank (2013); Hutton and Varughese (2016)
Discount rate (USD currency financial flows)	1.2%	%	OMB (2015)
% cost for construction (labour, supervision, transport) - % of material list cost	50%	%	BORDA, personal communication (2016)
% cost for overall project management, planning and administration (% of total construction)	25%	%	BORDA, personal communication (2016)
% cost for technical planning and engineering design (% of total construction)	5%	%	BORDA, personal communication (2016)

[a] Databank code: Inflation, GDP deflator (annual %). From: http://data.worldbank.org/indicator/NY.GDP.DEFL.KD.ZG
[b] Databank code: Purchasing power parity (PPP). From: http://data.worldbank.org/indicator/PA.NUS.PPP
[c] Additional cost for project management and technical planning not included for cost of on-site sanitation infrastructure.

Decentralized sewered system, India: LCA model inputs and findings

Table B.45: Construction materials and processes: Decentralized sewered, India[a]

Item[b, c]	Unit	Quantity
[C1 & C2] - Sewer line piping (total for 884m piping)		
Site clearance and leveling	35	m3
Excavation up to 1.5m depth	354	m3
Excavation up to 3m depth	0	m3
Backfilling and carefully compacting in layers	354	m3
Disposal of surplus soil	0	m3
PVC Pipe	2117	kg
Site clearance and leveling	35	m3
[C1 & C2] - Sewer line manholes (total for 47 units)		
Site clearance and leveling	47	m3
Excavation up to 1.5m depth	71	m3
Excavation up to 3m depth	0	m3
Backfilling and carefully compacting in layers	71	m3
Disposal of surplus soil	71	m3
Cement	25212	kg
Sand	50001	kg
Aggregate	65988	kg
Reinforcing steel	406	kg
[T1] - Biogas digester, dome construction (per unit) (Volume	*= 40m3) (2 Units)*	
Site clearance and leveling	63	m3
Excavation up to 1.5m depth	62	m3
Excavation up to 3m depth	145	m3
Backfilling and carefully compacting in layers	61	m3
Disposal of surplus soil	163	m3
Cement	10391	kg
Sand	51186	kg
Aggregate	47484	kg
Reinforcing steel	1103	kg
Galvanized steel	3	kg
[T2] - Anaerobic baffled reactor (per unit) (1 Unit)		
Site clearance and leveling	45	m3
Excavation up to 1.5m depth	181	m3
Excavation up to 3m depth	191	m3
Backfilling and carefully compacting in layers	35	m3
Disposal of surplus soil	294	m3
Cement	29266	kg
Sand	138163	kg
Aggregate	81972	kg
Reinforcing steel	2000	kg
PVC Pipe	444	kg
[T3] - Horizontal planted gravel filter (per unit) (12m x 18m x 0.6m depth) (1 Units)		
Site clearance and leveling	221	m3
Excavation up to 1.5m depth	177	m3
Excavation up to 3m depth	0	m3
Backfilling and carefully compacting in layers	221	m3
Disposal of surplus soil	221	m3
Cement	6911	kg
Sand	48073	kg
Aggregate	232529	kg

Reinforcing steel	197	kg
PVC Pipe	51	kg
HDPE liner	473	kg
[D/R] - Biogas reuse infrastructure (total for 2 institutional connections)		
Site clearance and leveling	5	m3
Excavation up to 1.5m depth	23	m3
Backfilling and carefully compacting in layers	23	m3
Galvanized steel piping	273	kg
Steel (cold rolled steel)	36	kg
Aluminum (stove burner head)	3	kg
Rubber hose pipe (PVC)	9	kg

[a] All data from CDD (2016) unless otherwise specified
[b]Assume density of loose soil = 1200kg/m3
[c]Assume density of crushed stone and sand = 1600 kg/m3

Table B.46: Land area: Decentralized sewered, India[a]

Item	Land area (m2)	Number of units	Total land area (m2)
[C1 & C2] - Sewer line piping	N/A	-	-
[T1] - Biogas digester (*per* unit) (Volume = 40m3) (2 Units)	68	2	135
[T2] - Anaerobic baffled reactor (*per* unit) (120m3) (1 Unit)	115	1	115
[T3] -Horizontal planted gravel filter (1 Unit)	265	1	265
[D/R] – Biogas piping & stoves	N/A	-	-
TOTAL LAND AREA			515

[a] All data from CDD (2016) unless otherwise specified

Table B.47: System and site description: Decentralized sewered, India

Item	Figure	Units	Reference
Year of project construction	2004	Year	CDD (2016)
Number of people served	575	people	Reynaud and Buckley, 2015
Number of households (wastewater connections)	120	households	Reynaud, 2014
Number of people *per* household	5	people / HH	Calculated
Wastewater generation *per capita*	29	LPCD	Reynaud and Buckley, 2015
COD generation *per-capita*	53.7	gPCD	Calculated using data from Reynaud and Buckley (2015) and Miller (2011)
Nitrogen generation *per-capita*	5.0	gPCD	Reynaud and Buckley, 2015
Site elevation	914.0	m	FloodMap.net (2014)
Site mean annual temperature	22.0	°C	World Bank (2012)
Water source: borehole (depth, m)	225	m	CDD (2016)
Water supply energy factor (borehole)	1.28	kWh/m3	Calculated based on borehole depth; CDD (2016)
Water distribution: tanker truck - energy factor	3.7	MJ/tkm	SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"
Water distribution: tanker truck - emissions factor	0.25	$kgCO_2e/tkm$	SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"

Table B.48: Wastewater, chemical oxygen demand (COD) inputs and outputs *per* SSC functional group: Decentralized sewered, India

Item	**Figure**	**Units**	**Reference**
Mean wastewater flow *per* day	16.5	m^3/day	Reynaud and Buckley, 2015
Total biodegradable organic waste (TOW)	30.9	kg COD/day	Calculated from Reynaud and Buckley, 2015
Total organic nitrogen loading	2.9	kg N/day	Calculated from Reynaud and Buckley, 2015
$COD_{(in,T1)}$	1871	mg COD/L	Calculated w/ data from Reynaud and Buckley (2015), Miller (2011)
$COD_{(removed,T1)}$	1358	mg COD/L	Calculated using data from Reynaud and Buckley (2015); Miller (2011)
$COD_{(out,T1)}$ and $COD_{(in,T2)}$	513	mg COD/L	Reynaud and Buckley (2015)
$COD_{(removed,T2)}$	193	mg COD/L	Reynaud and Buckley (2015)
$COD_{(out,T2)}$ and $COD_{(in,T3)}$	320	mg COD/L	Reynaud and Buckley (2015)
$COD_{(removed,T3)}$	214	mg COD/L	CDD (2014)
$COD_{(out,T3)\ (\text{*final discharge to stream})}$	106	mg COD/L	CDD (2014)

Table B.49: CH_4 and N_2O production *per* SSC functional group: Decentralized sewered, India

Item	**Figure**	**Units**	**Reference**
[T1]: Biogas digester			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: "Anaerobic reactor"
CH_4 production, gas phase (T1)	4.8	kg CH_4/day	Laramee et al. (2017)
CH_4 production, liquid phase (T1)	0.2	kg CH_4/day	Laramee et al. (2017)
Methane content	77%	%	Calculated
[T2]: Anaerobic baffled reactor			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: "Anaerobic reactor"
CH_4 production, gas phase (T2)	0.6	kg CH_4/day	Calculated
CH_4 production, liquid phase (T2)	0.1	kg CH_4/day	Calculated
Methane content	84%	%	Calculated
T3: Planted gravel filter (horizontal subsurface flow (HSSF)			
Methane correction factor (MCF)	0.1	-	IPCC (2013): Table 6.4: "Constructed wetland: HSSF"
CH_4 production, gas phase (T2)	0.06	kg CH_4/day	Calculated
CH_4 production, liquid phase (T2)	0.03	kg CH_4/day	Calculated
Methane content	80%	%	Calculated
Default emission factor (EF T-N_2O)	0.0079	kg N_2O/kg N	IPCC (2013): Table 6.7: "Constructed wetland: HSSF"
N_2O production *per* day	0.022	kg N_2O/day	Calculated (see methods)
D/R: Disposal (final discharge)			
Methane correction factor (MCF)	0.1	-	IPCC (2006): Table 6.3: "Sea, river and lake discharge"
CH_4 production, gas phase (D/R)	0.02	kg CH_4/day	Calculated
CH_4 production, liquid phase (D/R)	0.03	kg CH_4/day	Calculated
Methane content	64%	%	Calculated
Default emission factor (EF T-N_2O)	0.005	kg N_2O/kg N	IPCC (2006): Table 6.11: "Wastewater discharged into aquatic environments"
N_2O production *per* day	0.009	kg N_2O/day	Calculated (see methods)

Table B.50: Infrastructure and equipment replacement, recurrent materials, maintenance and transport: Decentralized sewered, India

Item	Figure	Units	Reference
Biogas piping and stove replacement [a]			
Energy use (total)	10,700	MJ	LCA analysis (present study)
Emissions impact (total)	900	$kgCO_2e$	LCA analysis (present study)
Recurrent materials [b]			
Cement: amount *per* year	50	kg/year	CDD (2016); SimaPro: "Cement, Portland {RoW} | market for | Alloc Def, S"
Personal protection equipment (boots and gloves) - PVC: amount *per* year	2.6	kg/year	CDD (2016); SimaPro: "Polyvinylchloride resin (S-PVC), suspension polymerisation, production mix, at plant, RER"
Work uniforms (coveralls) – cotton: amount *per* year	1.32	kg/year	CDD (2016); SimaPro: Textile, "woven cotton {GLO}|market for | Alloc Def, S"
Tools (pipe wrench, shovel, etc.) – steel: amount *per* year	6.6	kg/year	CDD (2016); SimaPro: Reinforcing steel {GLO}| market for | Alloc Def, S
Treatment infrastructure desludging (maintenance)			
T1: Biogas digester desludging: volume of sludge [c]	25	m3/ sludge year	CDD (2016); SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S
T1: Biogas digester desludging: transport to disposal site [d]	500	tonne-km / year	CDD (2016); SimaPro: SimaPro: Transport, truck <10t, EURO5, 80%LF, empty return
Other transport			
Vehicle travel (passenger vehicle, diesel) [e]	0	km / year	CDD (2016)

[a] See general assumptions for assumed lifespan. Cost *per capita per* year for infrastructure or equipment replacement is calculated as: Total impact (energy or emissions*number of replacement events)/(number of people served * analysis period).
[b] Recurrent material impacts calculated as: (material amount *per* year * energy or emission factor) / number of people served.
[c] Assume complete removal of sludge from T1 (75m3) occurs one time *per* 3 years.
[d] Assume: 12km roundtrip to disposal site*(75m3 sludge/3m3/trip)*(3tonne load + 2tonne truck)/3yrs

Table B.51: Conventional cooking fuels: Decentralized sewered, India

Item	Figure	Units	Reference
Conventional fuel used for cooking energy			
LPG (Liquid propane gas)	100%	%	Miller (2011); CDD 2016
Energy content of cooking fuels			
LPG: energy content	46.6	MJ/kg	https://www.ornl.gov
Methane: energy content	55.5	MJ/kg	Standard definition

Table B.52: Capital expenditure (CapEx): Decentralized sewered, India[a]

Item	Figure	Units	Reference
User CapEx			
[C1] - Household sewer line (total for 120 HH connections)	124,411	INR	CDD (2016)
Agency CapEx			
[C2] - Main sewer line piping and manholes	572,305	INR	CDD (2016)
[T1] - Biogas digesters (total for 2x 40m3 units)	1,038,805	INR	CDD (2016)
[T2] - Anaerobic baffled reactor (1 unit)	1,469,528	INR	CDD (2016)
[T3] - Horizontal planted filter (1 unit)	975,463	INR	CDD (2016)
[D/R] – Biogas piping and stoves (total for 2 institutional connections)	50,673	INR	CDD (2016)

[a] All costs adjusted to Year 2015 INR using GDP deflator and include estimated costs for planning and technical design.

Table B.53: Operational expenditures (OpEx): Decentralized sewered, India[a]

Item	Figure	Units	Reference
User OpEx - costs			
Water supply tariff (wastewater only) [b]	(372)	INR/cap/yr	Calculated based on CDD (2016)
Sanitation fixed charge	(35)	INR/cap/yr	Estimated based on IBNET (2016)
Agency OpEx - costs			
Human resource costs	(182)	INR/cap/yr	CDD (2016)
Material costs	(15)	INR/cap/yr	CDD (2016)
Energy and fuel costs	0	INR/cap/yr	CDD (2016)
Agency OpEx - benefits			
Fixed charge	35	INR/cap/yr	Estimated based on IBNET (2016)
Biogas tariff (theoretical) [c]	0.65	INR/MJ biogas	CDD, 2016 (subsidized price)

[a] All costs and benefits are reported in Year 2015 INR (adjusted using GDP deflator)
[b] Cost of water supply calculated according to wastewater generation *per capita* with 56% of daily water use from tanker (58 INR/m3) and 44% from community borehole (7 INR /m3 for electricity)
[c] Theoretical biogas tariff calculated *per* energy content, fuel efficiency and cost of conventional fuels (Laramee et al., 2017)

Table B.54: Capital maintenance expenditures (CapManEx): Decentralized sewered, India[a]

Item	Figure	Units	Reference
User CapManEx			
None	-	-	-
Agency CapManEx			
Desludging digesters and ABR	(113)	INR/cap/3yrs	CDD (2016)
PGF filter cleaning	(226)	INR/cap/5yrs	CDD (2016)
Gas stove and piping replacement	(88)	INR/cap/6yrs	CDD (2016)

[a] All costs and benefits are reported in Year 2015 INR (adjusted using GDP deflator)

Table B.55: Energy use findings by functional group: Infrastructure, use phase and net impacts: Decentralized sewered, India

Item	**Containment (MJ/*capita*/ year)**	**Conveyance (MJ/*capita*/ year)**	**Treatment (MJ/*capita*/ year)**	**Disposal / Reuse (MJ/*capita*/ year)**	**Net (MJ/*capita*/ year)**
Infrastructure	0.0	30.1	64.3	0.9	95.4
Use phase	0.0	133.4	7.3	-137.1	3.6
Net (mean)	0.0	163.4	71.7	-136.2	99.0
Lower bound uncertainty	0.0	126.0	60.3	-173.4	12.8
Upper bound uncertainty	0.0	213.7	92.4	-98.9	207.3

Table B.56: Global warming potential (GWP) findings by functional group: Infrastructure, use phase and net impacts: Decentralized sewered, India

Item	**Containment (kgCO_2e/ *capita*/year)**	**Conveyance (kgCO_2e/ *capita*/year)**	**Treatment (kgCO_2e/ *capita*/year)**	**Disposal / Reuse (kgCO_2e/ *capita*/year)**	**Net (kgCO_2e/ *capita*/year)**
Infrastructure	0.0	3.7	9.4	0.1	13.2
Use phase	0.0	18.7	34.8	-7.0	46.5
Net (mean)	0.0	22.5	44.1	-6.9	59.7
Lower bound uncertainty	0.0	17.0	34.6	-6.9	44.8
Upper bound uncertainty	0.0	28.8	57.4	-6.2	80.0

Table B.57: Infrastructure phase energy use and global warming potential (GWP) by source of impact: Decentralized sewered, India

Item	Energy use (MJ/*capita*/ year)	Percentage of energy use [a]	GWP ($kgCO_2e$/ *capita*/year)	Percentage of GWP (%) [a]
Cement (I)	41.7	17%	9.8	7%
Steel (I)	10.6	4%	1.0	1%
Clay bricks (I)	0.0	0%	0.0	0%
PVC Pipe / plastics (I)	20.1	8%	0.8	1%
Transportation (I)	14.3	6%	1.0	1%
Construction works (I)	7.7	3%	0.5	0%
Other (I)	1.0	0%	0.1	0%

[a] Percentage of lifecycle energy use or GWP impacts excluding reductions in energy use or GWP due to recovered CH_4 or co-product offsets

Table B.58: Use phase energy by source of impact: Decentralized sewered, India

Item	Energy use (MJ/*capita*/year)	Percentage of energy use (%) [a]
EE,j - Electricity use (U)	0.0	0%
EM,j - Material use (U)	4.6	2%
EW,j - Water supply (U)	133.4	56%
EV,j - Vehicle use (U)	5.5	2%
EH,j - Human energy (U)	0.0	0%
EI,j - Co-products (U)	-139.9	-

[a] Percentage of lifecycle energy use excluding reductions due to co-product offsets

Table B.59: Use phase global warming potential (GWP) by source of impact: Decentralized sewered, India

Item	Energy use (MJ/*capita*/year)	Percentage of energy use [a]
CE,j - Electricity inputs (U)	0.0	0%
CM,j - Material inputs (U)	0.4	0%
CW,j - Water supply (U)	18.7	14%
CV,j - Vehicle use (U)	0.4	0%
CP-CH_4,j - Treatment (CH_4) (U)	93.7	71%
CP-N_2O,j - Treatment (N_2O) (U)	5.8	4%
CR,j - Recovered CH_4 (U)	-63.0	-
CC,j - Co-products (U)	-9.6	-

[a] Percentage of lifecycle GWP excluding reductions due to recovered CH_4 or co-product offsets

Table B.60: User equivalent annual cost (EAC): Decentralized sewered, India

Item	**User EAC - mean (USD/*capita*/year)** [a]	**User EAC - lower bound (USD/*capita*/year)** [a]	**User EAC - upper bound (USD/*capita*/year)** [a]
CapEx	$(1)	$(1)	$(1)
OpEx - Costs	$(24)	$(35)	$(18)
OpEx - Benefits	$-	$-	$-
CapManEx	$-	$-	$-
Carbon financing	$-	$-	$-
Net User EAC	$(25)	$(36)	$(18)

[a] All costs in Year 2015 USD

Table B.61: Agency equivalent annual cost (EAC): Decentralized sewered, India

Item	**Agency EAC - mean (USD/*capita*/year)** [a]	**Agency EAC - lower bound (USD/*capita*/year)** [a]	**Agency EAC - upper bound (USD/*capita*/year)** [a]
CapEx	$(34)	$(60)	$(21)
OpEx - Costs	$(12)	$(16)	$(8)
OpEx - Benefits	$6	$6	$6
CapManEx	$(5)	$(6)	$(4)
Carbon financing	$-	$-	$-
Net Agency EAC	$(44)	$(75)	$(27)

[a] All costs in Year 2015 USD

Non-sewered system, India: LCA model inputs and findings

Table B.62: Construction materials and processes: Non-sewered, India[a]

Item[b, c]	Quantity	Unit
[C/S] - Pit substructure (single pit with concrete ring; 2m depth) (per unit)		
Site clearance and leveling	1.2	m2
Excavation up to 3m depth	2.5	m3
Ground leveling after excavation	1.2	m2
Backfilling and carefully compacting in layers	1.6	m2
Disposal of surplus soil	0.9	m3
Cement	152	kg
Sand	331	kg
Aggregate	727	kg
Reinforcing steel	14.4	kg
PVC pipe	3	m
[C1] – Primary conveyance: Vacuum tanker		
Vacuum tanker; 4tonne load capacity (fulltime use) - SimaPro (assume '16ton lorry')	1	unit
[C2] - PVC piping and registers (between treatment units)		
Excavation up to 1.5m depth	34.5	m3
Soling (rubble fill)	5.2	m3
Cement	3195	kg
Sand	7530	kg
Aggregate	11310	kg
PVC piping	62	kg
[T1] - Biogas digesters, stabilization tank and ancillary infrastructure		
Ramp to feeding tank (T1)		
Excavation up to 1.5m depth	9.5	m3
1.2m depth fill material	85	m3
Cement	2892	kg
Sand	6549	kg
Aggregate	14379	kg
Feeding tank – prefab (T1)		
Excavation up to 1.5m depth	2.3	m3
Cement	8635	kg
Sand	19393	kg
Aggregate	41182	kg
Fiberglass prefab	700	kg
Biogas digester – prefab (per unit) (2 Units)		
Excavation up to 3m depth	4	m3
Soling (rubble fill)	1.3	m3
Backfill	4	m3
Cement	78	kg
Sand	335	kg
Aggregate	740	kg
Fiberglass prefab	500	kg
Stabilization tank - prefab (per unit) (2 units)		
Excavation up to 3m depth	3.6	m3
Soling (rubble fill)	3.1	m3
Backfill	3.6	m3
Cement	186	kg
Sand	804	kg
Aggregate	1776	kg

Fiberglass prefab	1100	kg
Registers at stabilization tank (per unit) (2 units)		
Excavation up to 1.5m depth	4.6	m3
Soling (rubble fill)	3.1	m3
Cement	1158	kg
Sand	3088	kg
Aggregate	4426	kg
Excavation up to 1.5m depth	4.6	m3
Soling (rubble fill)	3.1	m3
[T2-S] – Secondary treatment (solids fraction): Sludge drying beds (solids)		
Sludge drying beds (all units) (T3)		
Excavation up to 1.5m depth	290	m3
Soling (rubble fill)	86	m3
Cement	17191	kg
Sand	96086	kg
Aggregate	188200	kg
PVC piping	338	kg
Polycarbon roofing	363	kg
Steel (roof structure)	2385	kg
[T2-L] – Secondary treatment (liquid fraction): anaerobic baffled reactor; planted gravel filter		
Anaerobic baffled reactor - prefab (1 unit)		
Excavation up to 3m depth	18	m3
Soling (rubble fill)	2.4	m3
Backfill	18	m3
Cement	186	kg
Sand	804	kg
Aggregate	1776	kg
Fiberglass prefab	1100	kg
Planted gravel filter (1 Unit)		
Excavation up to 1.5m depth	15.9	m3
Soling (rubble fill)	4.6	m3
Cement	2550	kg
Sand	6199	kg
Aggregate	27253	kg
PVC piping	14	kg
[D/R] – Collection tank; percolation pit; gas piping and stove		
Collection Tank (D/R)		
Excavation	18	m3
Soling (rubble fill)	0.6	m3
Cement	1111	kg
Sand	4650	kg
Aggregate	9006	kg
PVC piping	103	kg
Percolation pit		
Excavation	125	m3
Soling (rubble fill)	23	m3
Cement	1878	kg
Sand	50206	kg
Aggregate	165128	kg
PVC piping	66	kg
Biogas piping and stove (per connection) (2 connections)		
Site clearance	1.5	m2
Excavation up to 1.5m depth	7.5	m3
Backfilling and carefully compacting in layers	7.5	m3
Galvanized steel piping	94	kg

Steel (cold rolled steel)	2.4	kg
Aluminum (stove burner head)	0.2	kg
Rubber hose pipe (PVC)	0.6	kg

[a] All data from CDD (2016) unless otherwise specified
[b] Assume density of loose soil = 1200kg/m3
[c] Assume density of crushed stone and sand = 1600 kg/m3

Table B.63: Land area: Non-sewered, India[a]

Item	Land area (m2)
[C/S] - Household latrines	0
[C1] - Vacuum tanker	0
[C2] - Waste pipes at treatment plant	0
[C3] - N/A	0
[T1] - Biogas digesters and supporting infrastructure (ramp)	200
[T2] – Anaerobic baffled reactor, stabilization tanks	200
[T3] – Planted gravel filter; Sludge drying beds	225
[D/R] – Percolation pit, composting, gas	110
TOTAL LAND AREA	735

[a] All data from CDD (2016) unless otherwise specified

Table B.64: System and site description: Non-sewered, India

Item	Figure	Units	Reference
Year of project construction	2015	Year	CDD (2016)
Mean TS of fecal sludge	4%	%	CDD (2016)
Mean kg FS input *per* day (wet wt)	1978	kg/day	CDD (2016)
Mean kg FS input *per* day (dry wt)	83	kg/day	Calculated
Total organic solids produced *per-capita per*-day	39	g/cap/day	Rose et al. (2016)
Depth of pit emptied	1m	m	CDD (2016)
Number of people served	6887	people	Calculated
Mean number of people sharing toilet [a]	4.4	people/latrine	CDD (2016)
Number of toilets	1565	latrines	Calculated
Wastewater generation *per capita*	7	LPCD	CDD (2016); Mara (1981)
COD generation *per-capita* [b]	53.7	gPCD	Calculated using data from Reynaud and Buckley (2015) and Miller (2011)
Nitrogen generation *per-capita*	5.0	gPCD	Reynaud and Buckley (2015)
Site elevation	914	m	FloodMap.net (2014)
Site mean annual temperature	22	°C	World Bank (2012)
Borehole water source (depth, m)	225	m	CDD (2016)
Energy req'd for water supply (borehole)	1.28	kWh/m3	Calculated based on borehole depth and assumed pump efficiencies
Water distribution: tanker truck - energy factor	3.7	MJ/tkm	SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"
Water distribution: tanker truck - emissions factor	0.25	$kgCO_2e/tkm$	SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"

[b] *Per capita* COD assumed to be equivalent to COD *per capita* measured for sewered system (no data available specific to site)

Table B.65: Chemical oxygen demand (COD) inputs and outputs *per* SSC functional group: Non-sewered, India

Item	Figure	Units	Reference
Total organic waste (TOW), COD(in,C/S)	312	kg COD/day	Calculated (see methods)
Total organic nitrogen loading	34	kg N / day	Calculated
$COD_{(removed,C/S)}$	276	kg COD / day	Calculated [a]
$COD_{(in,T1)}$	37	kg COD / day	Calculated
$COD_{(removed,T1)}$	2	kg COD / day	Calculated based on measured biogas production at T1
$COD_{(in,T2)}$	34	kg COD / day	Calculated
$COD_{(removed,T2)}$	30	kg COD / day	Calculated based on Koottatep et al. (2005)
$COD_{(in, D/R)}$ (final discharge to land application)	4	kg COD / day	Calculated
$COD_{(removed,D/R)}$	4	kg COD / day	Calculated (all COD assumed to be removed)

[a] Calculated based on mean and range of kg fecal sludge inputs to treatment plant *per* day, total solids content (%) and dry weight of organic solids produced *per capita per* day (see methods).

Table B.66: CH_4 and N_2O production *per* functional group: Non-sewered, India

Item	Figure	Units	Reference
[C/S]: Household pit latrine (or septic tank)			
Methane correction factor (MCF)	0.7	-	IPCC (2006): Chapter 6, Table 6.3: "latrine: flush water use"
Biogas production (gas phase)	48.4	kg CH_4/day	Calculated (see methods)
Methane content	71%	%	Calculated (see methods)
[T1]: Biogas digester			
Methane correction factor (MCF)	0.9	-	IPCC (2006): Table 6.3: “Anaerobic reactor”
CH_4 production, gas phase (T1)	0.4	kg CH_4/day	CDD (2016)
Methane content	71%	%	Calculated (see methods)
[T2-S]: Sludge drying beds			
Methane correction factor (MCF)	0.2	-	IPCC (2006): Table 6.3: "Anaerobic shallow lagoon"
Biogas production (gas phase)	1.6	kg CH_4/day	Calculated (see methods)
Methane content	71%	%	Calculated (see methods)
Default emission factor (EF T-N_2O)	0.005	kg N_2O/ kg N	IPCC (2006): Chapter 10 (Solid storage)
N_2O production *per* day	0.172	kg N_2O/day	Calculated (see methods)

[a] Assume methane production in liquid phase negligible COD concentration > 5000mg/L

Table B.67: Conveyance: Water supply, motorized and manual conveyance inputs: Non-sewered, India

Item	Figure	Units	Reference
[C/S]: Household pit latrine (or septic tank) inputs			
Water supply: borehole [a]	48.2	m3/day	Calculated
Water distribution: piped (gravity)	31.0	m3/day	Calculated; CDD (2016)
Water distribution: tanker truck	68.9	t-km/day	Calculated; CDD (2016)
C1: Primary Conveyance - Inputs			
Water inputs: borehole [b]	0.9	m3/day	CDD (2016)
Pumping / desludging (on-site sanitation) [c]	690	m3 sludge/ year	CDD (2016); SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S
Vehicle – light duty (<10tonne) [d]	0	tonne-km/year	CDD (2016); SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"
Vehicle – heavy duty (10-20tonne) [d]	19972	tonne-km/year	CDD (2016); SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"
Human power - low intensity (e.g. cleaning) e	520	Person-hrs/year	CDD (2016)
Human power - low intensity (e.g. walking)	0	Person-hrs/year	CDD (2016)
Human power - medium intensity (e.g. digging) f	1040	Person-hrs/year	CDD (2016)
Human power - high intensity (e.g. pushing wheelbarrow)	0	Person-hrs/year	CDD (2016)
C1: Secondary Conveyance - Inputs			
Human power - low intensity (e.g. cleaning) [g]	520	Person-hrs/year	CDD (2016)
Human power - low intensity (e.g. walking) [g]	520	Person-hrs/year	CDD (2016)
Human power - medium intensity (e.g. digging) [g]	520	Person-hrs/year	CDD (2016)
Human power - high intensity (e.g. pushing wheelbarrow) [g]	520	Person-hrs/year	CDD (2016)

[a] Pour flush latrines primarily used in study site
[b] Additional water used for cleaning tools and equipment at treatment system.
[c] Desludging of on-site pits performed with vacuum tanker
[d] Primary conveyance with heavy duty vacuum tanker
[e] Assumption: 1hr/day offloading/cleaning; 2 people; 1 site/day; 5days/week; 52 weeks/year
[f] Assumption: 2hr/day pit-emptying; 2 people; 1 site/day; 5days/week; 52 weeks/year
[g] Assumption: 1 operator: 2hr/day; 5days/week; 52 weeks/year

Table B.68: Infrastructure and equipment replacement, recurrent materials, maintenance and transport: Non-sewered, India

Item	Figure	Units	Reference
On-site sanitation (pit latrine) replacement [a]			
Energy use *per* latrine	1820	MJ/unit	Calculated [e]
Emissions *per* latrine	222	$kgCO_2e$/unit	Calculated [e]
Equipment and tools (annual replacement) (C1) a			
Energy use *per* year	0.15	MJ/year	Calculated [e]
Emissions impact *per* year	0.01	$kgCO_2e$/year	Calculated [e]
Biogas piping and stove replacement [a]			
Energy use (total)	6,027	MJ	Calculated [e]
Emissions impact (total)	504	$kgCO_2e$	Calculated [e]
Recurrent materials [b]			
Cement: amount *per* year	600	kg/year	CDD (2016); SimaPro: "Cement, Portland {RoW} | market for | Alloc Def, S"
Cleaning disinfectant: amount *per* year	12	kg/year	CDD (2016); SimaPro: Sodium hypochlorite, without water, in 15% solution state {GLO}|market for | Alloc Def, S
Personal protection equipment (boots and gloves) -PVC: amount *per* year	10.4	kg/year	CDD (2016); SimaPro: Polyvinylchloride resin (S-PVC), suspension polymerisation, production mix, at plant, RER
Work uniforms (coveralls) – cotton: amount *per* year	5.28	kg/year	CDD (2016); SimaPro: Textile, "woven cotton {GLO}|market for | Alloc Def, S"
Treatment infrastructure desludging (maintenance)			
T1: Treatment system desludging [c]	51	m3/sludge year	CDD (2016); SimaPro: Liquid manure spreading, by vacuum tanker {GLO}|market for | Alloc Def, S
T1: Treatment system desludging: transport to disposal site [d]	0	tonne-km/year	CDD (2016); Transport, truck < 10t, EURO5, 80%LF, empty return

[a] See general assumptions for assumed lifespan. Cost *per capita per* year for infrastructure or equipment replacement is calculated as: Total impact (energy or emissions*number of replacement events)/(number of people served * analysis period).
[b] Recurrent material impacts calculated as: (material amount *per* year * energy or emission factor) / number of people served.
[c] Assume removal of sludge 2x *per* year for: (Digester = 2*4m3) + (ABR = 1*14.4m3) + (ST = 2*14.4m3)
[d] Desludged directly to treatment site drying beds
[e] Calculated in SimaPro Version 8.3. See previous "Construction materials and processes" table for more information on material use.

Table B.69: Conventional cooking fuels: Non-sewered, India

Item	Figure	Units	Reference
Conventional fuel used for cooking energy			
Liquid propane gas (LPG)	100%	%	CDD (2016)
Energy content of cooking fuels			
LPG: energy content	46.6	MJ/kg	IEI, 2004
Methane: energy content	55.5	MJ/kg methane	Standard definition

Table B.70: Capital expenditure (CapEx): Non-sewered, India[a]

Item	Figure	Units	Reference
User CapEx			
[C/S] - Pit latrine (concrete ring substructure) (*per* unit) [b]	9,958	INR	CDD (2016)
Agency CapEx			
[C1] – Vacuum tanker	1,600,000	INR	CDD (2016)
[C2] – Piping and registers	234,938	INR	CDD (2016)
[T1] - Biogas digesters, stabilization tank and ancillary infrastructure	218,893	INR	CDD (2016)
[T2-S] – Secondary treatment (solids fraction): Sludge drying beds (solids)	1,914,107		
[T2-L] – Secondary treatment (liquid fraction): anaerobic baffled reactor; planted gravel filter	942,476	INR	CDD (2016)
[D/R] – Collection tank; percolation pit; gas piping and stove	448,154	INR	CDD (2016)

[a] All costs adjusted to Year 2015 INR using GDP deflator and include estimated costs for planning and technical design

[b] *Per-capita* cost of pit latrine calculated as cost *per* unit / mean number of people sharing pit latrine

Table B.71: Operational expenditure (OpEx): Non-sewered, India[a]

Item	Figure	Units	Reference
User OpEx - costs			
Water supply tariff (wastewater only)	149	INR/cap/yr	Calculated w/ data: CDD (2016)
Pit emptying fee [b]	34	INR/cap/yr	Calculated w/ data: CDD (2016)
Agency OpEx - costs			
Human resource costs	105	INR/cap/yr	Calculated w/ data: CDD (2016)
Material costs	30	INR/cap/yr	Calculated w/ data: CDD (2016)
Energy and fuel costs	16	INR/cap/yr	Calculated w/ data: CDD (2016)
Human resource costs	105	INR/cap/yr	Calculated w/ data: CDD (2016)
Agency OpEx - benefits			
Pit emptying revenue [b]	27	ZMW/cap/yr	Calculated w/ data: CDD (2016)
Biogas tariff (theoretical) [c]	0.65	ZMW/MJ biogas	CDD, 2016 (subsidized price)

[a] All costs and benefits are reported in Year 2015 INR (adjusted using GDP deflator)
[b] Pit emptying fee is assumed as a 'cost' to the user and a 'benefit' to the agency. Total revenue generated in 2016 is divided by the total number of users to calculate an average annual cost *per* user.
[c] Theoretical biogas tariff calculated *per* energy content, fuel efficiency and cost of conventional fuels (Laramee et al., 2017)

Table B.72: Capital maintenance expenditure (CapManEx): Non-sewered, India[a]

Item	Figure	Units	Reference
User CapManEx			
Pit latrine replacement (*per* unit) [b]	9958	INR / unit	CDD (2016)
Agency CapManEx			
Filter Material Replacement in SDB	2.2	INR / cap / 3yrs	CDD (2016)
Filter material cleaning and replacement in SDB	3.5	INR / cap / 3yrs	CDD (2016)
Cleaning filter and plantation of PGF	1.7	INR / cap / 3yrs	CDD (2016)
Desludging digesters (agency)	0.3	INR / cap / yr	CDD (2016)
Desludging ABR (agency)	0.2	INR / cap / yr	CDD (2016)
Desludging STs (agency)	0.3	INR / cap / yr	CDD (2016)
Replace gas stoves and piping (agency)	11.9	INR / cap / 6yrs	CDD (2016)

[a] All costs and benefits are reported in Year 2015 INR (adjusted using GDP deflator)
[b] An equal number of pit latrines are assumed to be replaced each year for the duration of the analysis period. Cost of pit latrine categorized as CapEx for assumed lifespan of pit and thereafter categorized as CapManEx. Annual *per-capita* cost for pit latrine replacement is calculated as: (unit cost of pit latrine) / (lifespan * mean number of people sharing pit latrine).

Table B.73: Energy use findings by functional group: Infrastructure, use phase and net impacts: Non-sewered, India

Item	**Containment (MJ/*capita*/ year)**	**Conveyance (MJ/*capita*/ year)**	**Treatment (MJ/*capita*/ year)**	**Disposal / Reuse (MJ/*capita*/ year)**	**Net (MJ/*capita*/ year)**
Infrastructure	20.7	2.6	6.2	0.6	30.0
Use phase	34.0	16.3	0.3	-0.8	49.7
Net (mean)	54.7	18.8	6.4	-0.2	79.7
Lower bound uncertainty	35.4	5.6	2.3	-0.7	42.6
Upper bound uncertainty	84.2	69.1	22.3	1.7	177.3

Table B.74: Global warming potential (GWP) findings by functional group: Infrastructure, use phase and net impacts: Non-sewered, India

Item	**Containment ($kgCO_2e$/ *capita*/year)**	**Conveyance ($kgCO_2e$/ *capita*/year)**	**Treatment ($kgCO_2e$/ *capita*/year)**	**Disposal / Reuse ($kgCO_2e$/ *capita*/year)**	**Net ($kgCO_2e$/ *capita*/year)**
Infrastructure	2.5	0.2	0.6	0.1	3.5
Use phase	69.2	5.4	2.8	5.4	82.8
Net (mean)	71.8	5.6	3.4	5.5	86.2
Lower bound uncertainty	37.5	5.6	4.0	5.4	52.5
Upper bound uncertainty	104.6	5.6	2.9	5.5	118.6

Table B.75: Infrastructure phase energy use and global warming potential (GWP) by source of impact: Non-sewered, India

Item	**Energy use (MJ/*capita*/ year)**	**Percentage of energy use [a]**	**GWP (kgCO_2e/ *capita*/year)**	**Percentage of GWP (%) [a]**
Cement (I)	10.0	12%	1.9	2%
Steel (I)	6.3	8%	0.6	1%
Clay bricks (I)	0.0	0%	0.0	0%
PVC Pipe / plastics (I)	6.1	8%	0.3	0%
Transportation (I)	2.8	3%	0.2	0%
Construction works (I)	1.6	2%	0.1	0%
Other (I)	3.3	4%	0.4	0%

[a] Percentage of lifecycle energy use or GWP impacts excluding reductions in energy use or GWP due to recovered CH_4 or co-product offsets

Table B.76: Use phase energy use by source of impact: Non-sewered, India

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use (%) [a]**
EE,j - Electricity use (U)	0.0	0%
EM,j - Material use (U)	9.9	12%
EW,j - Water supply (U)	25.3	31%
EV,j - Vehicle use (U)	14.8	18%
EH,j - Human energy (U)	0.6	1%
EI,j - Co-products (U)	-1.0	-

[a] Percentage of lifecycle energy use excluding reductions due to co-product offsets

Table B.77: Use phase global warming potential (GWP) by source of impact: Non-sewered, India

Item	**Energy use (MJ/*capita*/year)**	**Percentage of energy use [a]**
CE,j - Electricity inputs (U)	0.0	0%
CM,j - Material inputs (U)	1.8	2%
CW,j - Water supply (U)	4.1	5%
CV,j - Vehicle use (U)	4.9	6%
CP-CH_4,j - Treatment (CH_4) (U)	67.1	77%
CP-N_2O,j - Treatment (N_2O) (U)	5.4	6%
CR,j - Recovered CH_4 (U)	-0.4	-
CC,j - Co-products (U)	-0.1	-

[a] Percentage of lifecycle GWP excluding reductions due to recovered CH_4 or co-product offsets

Table B.78: User equivalent annual cost (EAC): Non-sewered, India

Item	**User EAC - mean (USD/*capita*/year) [a]**	**User EAC - lower bound (USD/*capita*/year) [a]**	**User EAC - upper bound (USD/*capita*/year) [a]**
CapEx	$(5)	$(7)	$(4)
OpEx - Costs	$(11)	$(21)	$(5)
OpEx - Benefits	$-	$-	$-
CapManEx	$(2)	$(3)	$(1)
Carbon financing	$-	$-	$-
Net User EAC	$(18)	$(32)	$(10)

[a] All costs in Year 2015 USD

Table B.79: Agency equivalent annual cost (EAC): Non-sewered, India

Item	**Agency EAC - mean (USD/*capita*/year) [a]**	**Agency EAC - lower bound (USD/*capita*/year) [a]**	**Agency EAC - upper bound (USD/*capita*/year) [a]**
CapEx	$(6)	$(30)	$(2)
OpEx - Costs	$(9)	$(30)	$(3)
OpEx - Benefits	$2	$4	$1
CapManEx	$(0)	$(1)	$(0)
Carbon financing	$-	$-	$-
Net Agency EAC	$(14)	$(57)	$(4)

[a] All costs in Year 2015 USD

B.2 Sensitivity analysis: Energy, GWP, user and agency costs

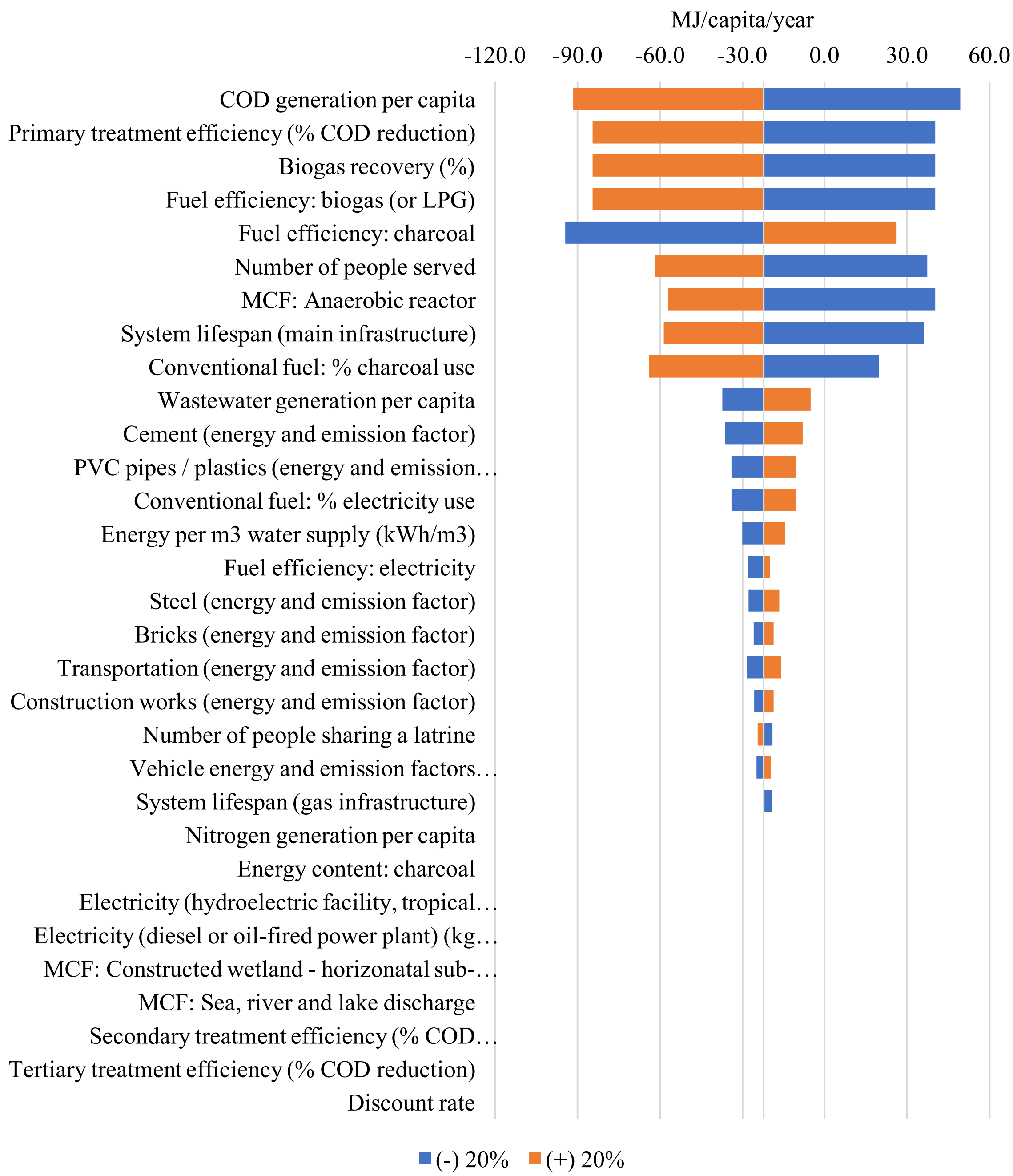


Figure B.1: Sensitivity analysis: Energy use: Decentralized Sewered, Zambia

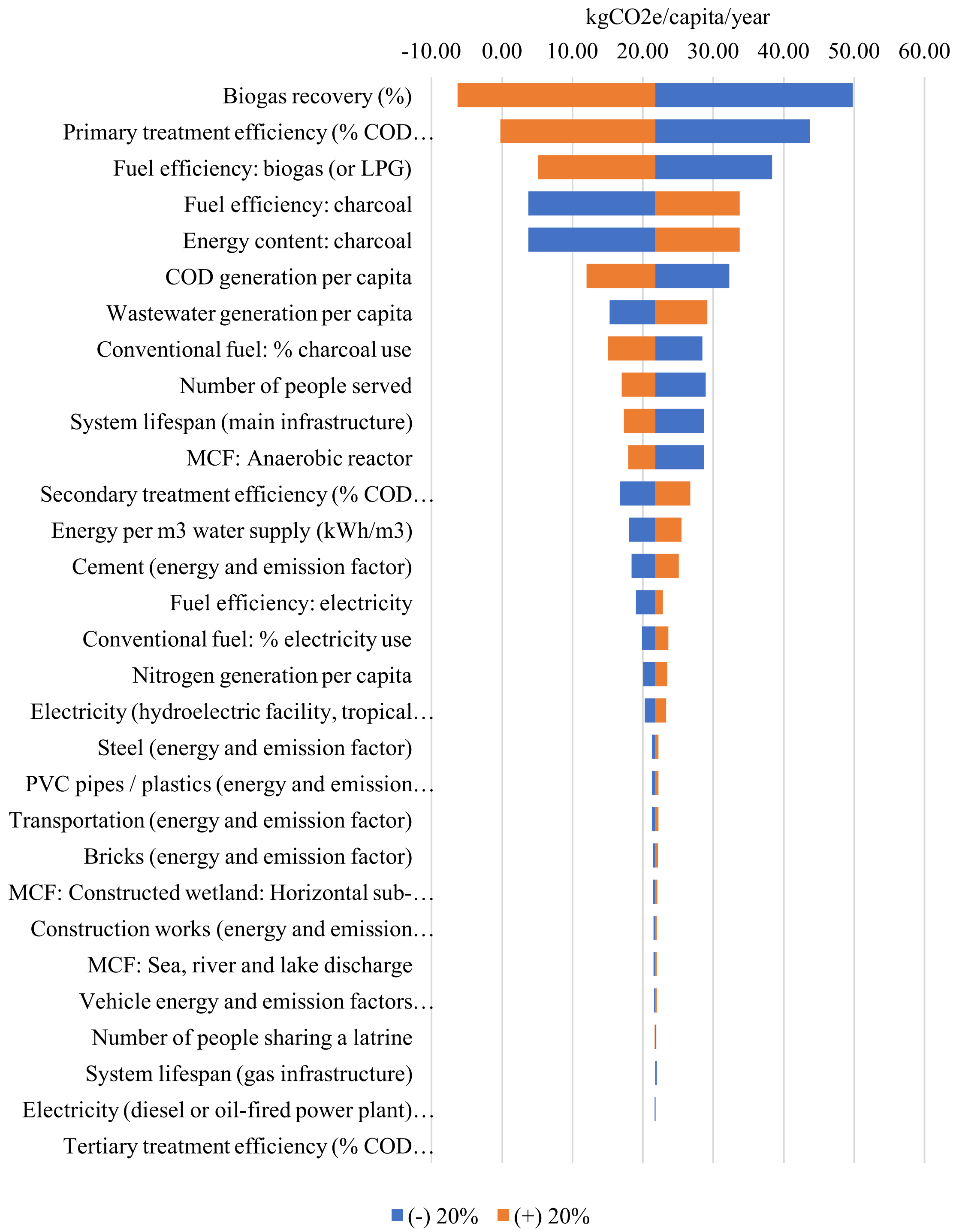


Figure B.2: Sensitivity analysis: Global Warming Potential (GWP): Decentralized Sewered, Zambia

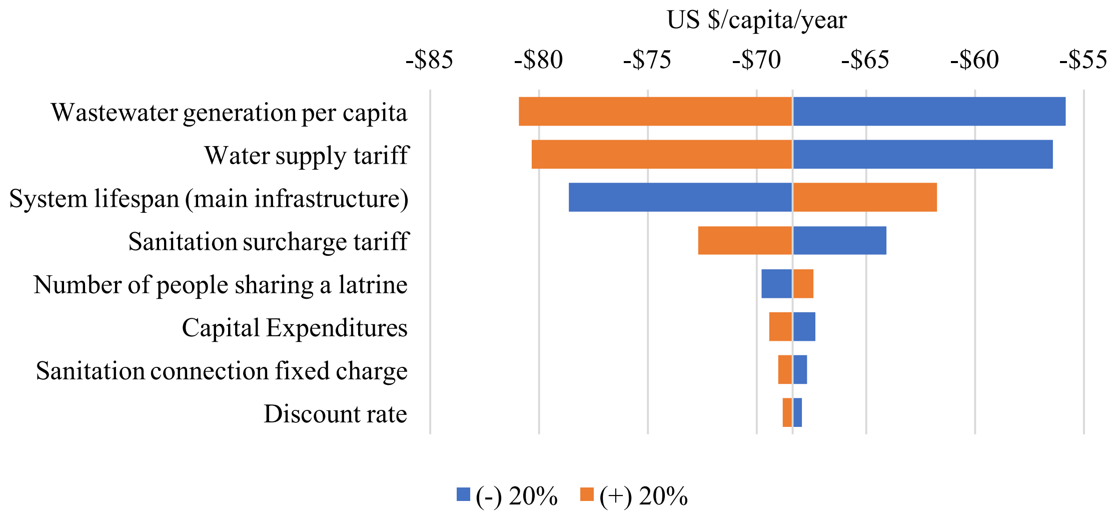


Figure B.3: Sensitivity analysis: Equivalent annual cost (EAC), User: Decentralized Sewered, Zambia

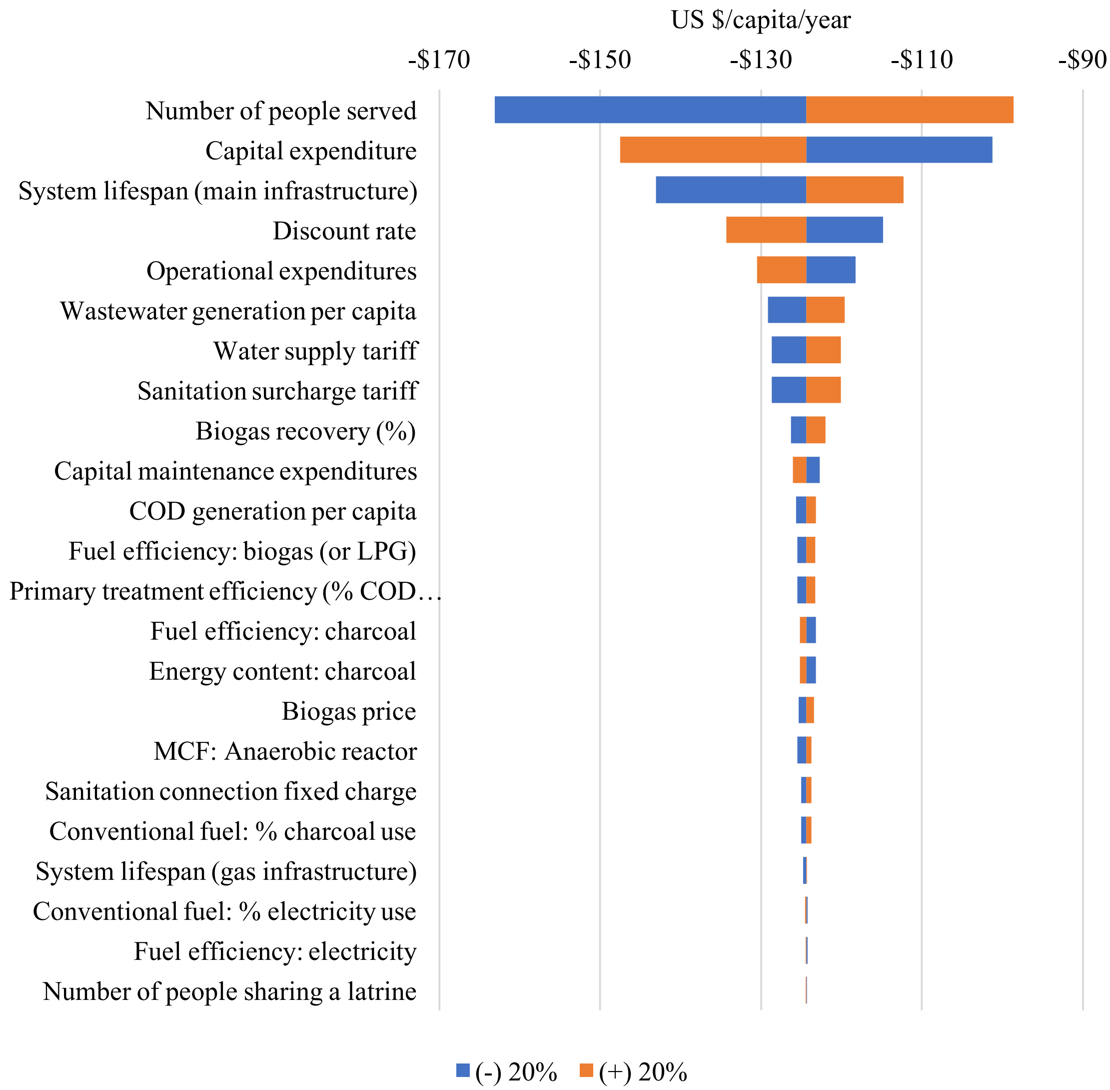


Figure B.4: Sensitivity analysis: Equivalent annual cost (EAC), Agency: Decentralized Sewered, Zambia

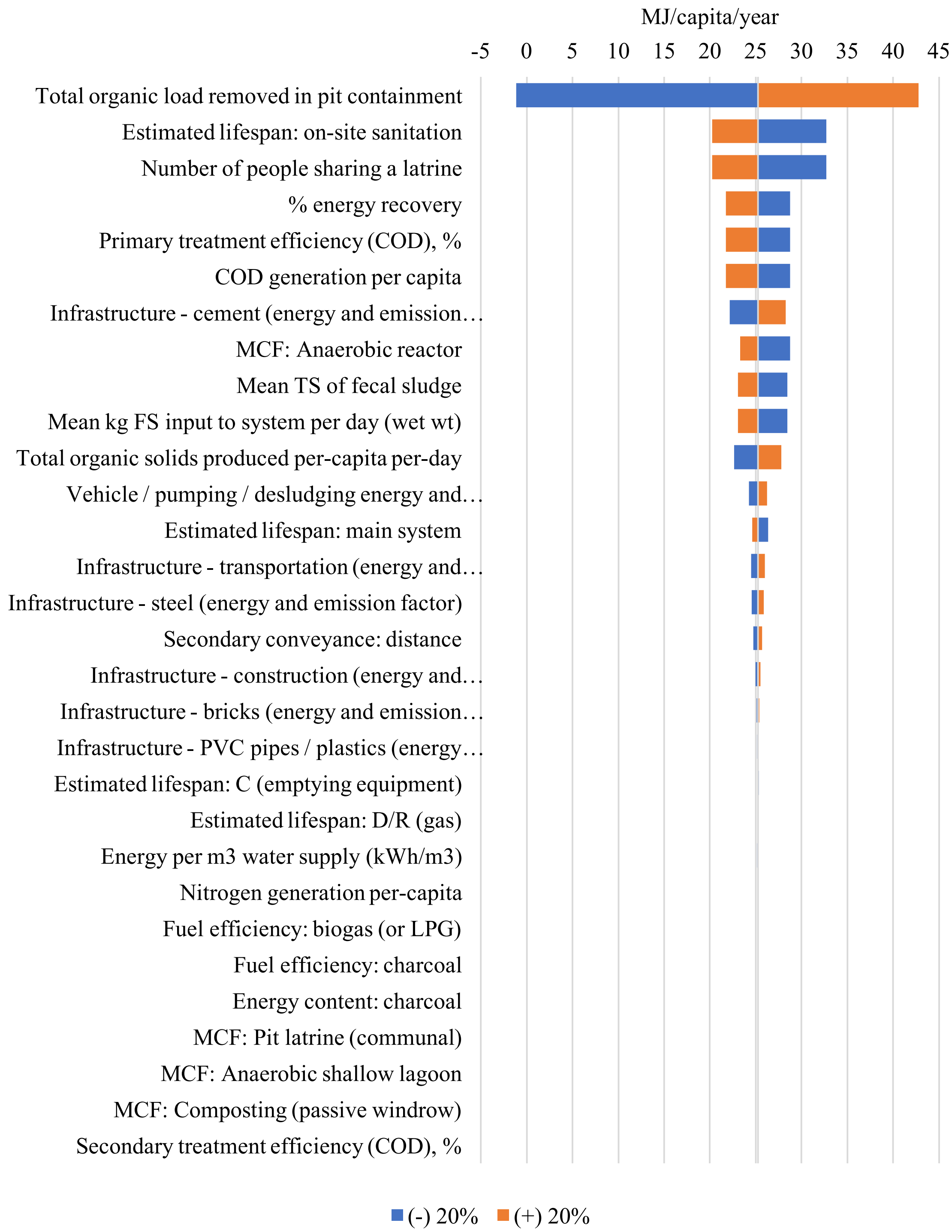


Figure B.5: Sensitivity analysis: Energy use: Non-sewered, Zambia

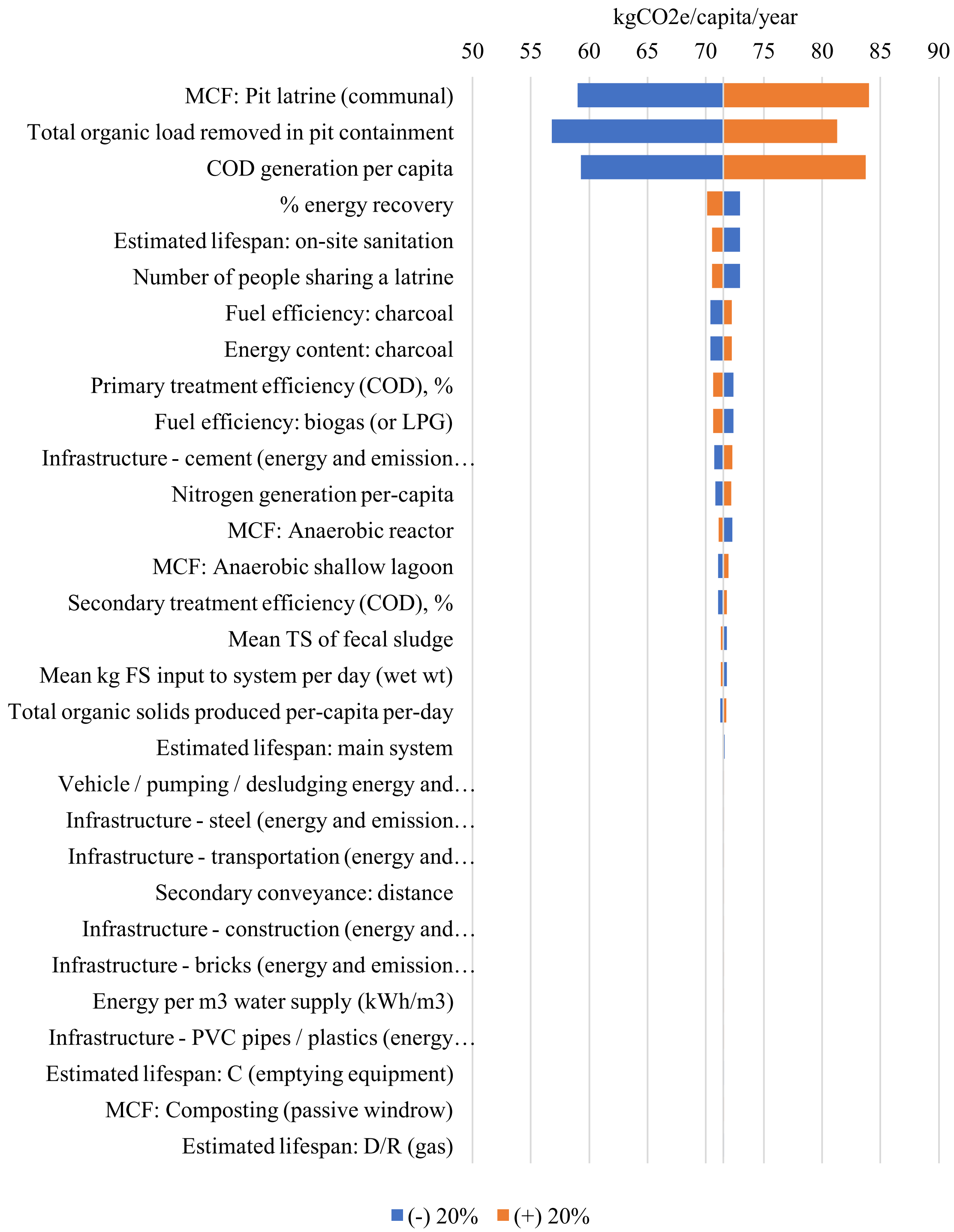


Figure B.6: Sensitivity analysis: Global Warming Potential (GWP): Non-sewered, Zambia

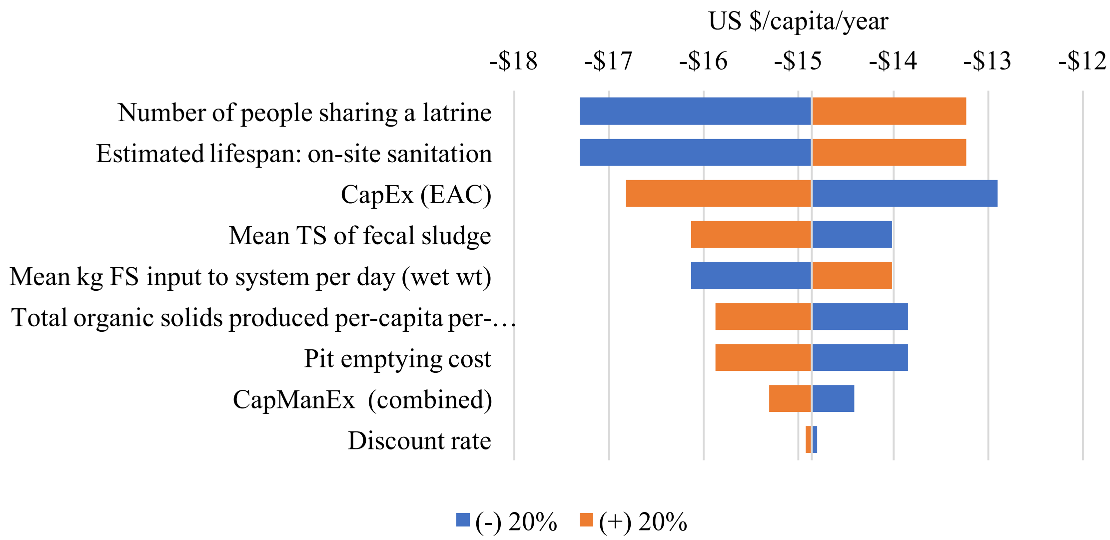


Figure B.7: Sensitivity analysis: Equivalent annual cost (EAC), User: Non-sewered, Zambia

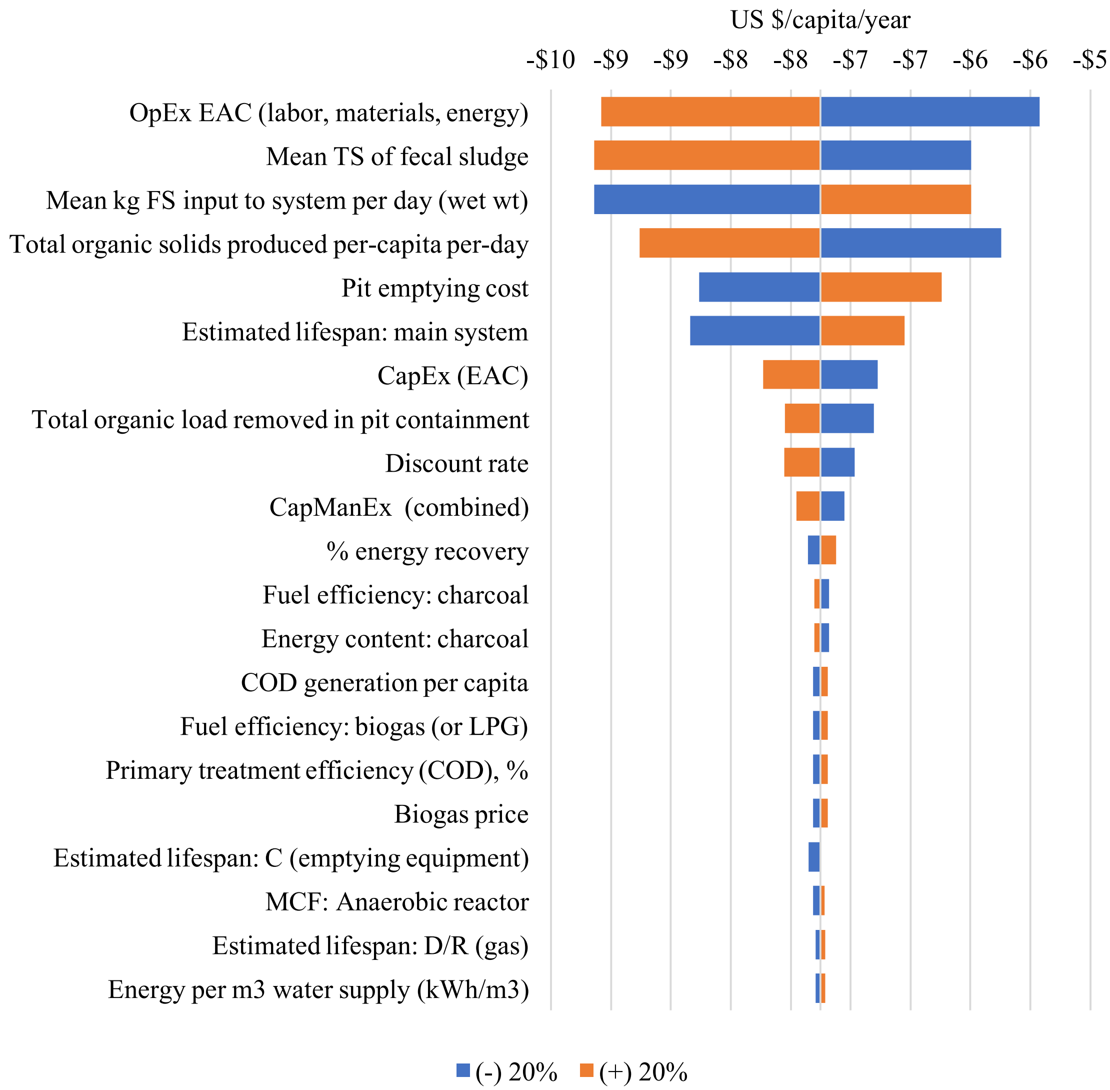


Figure B.8: Sensitivity analysis: Equivalent annual cost (EAC), Agency: Non-sewered, Zambia

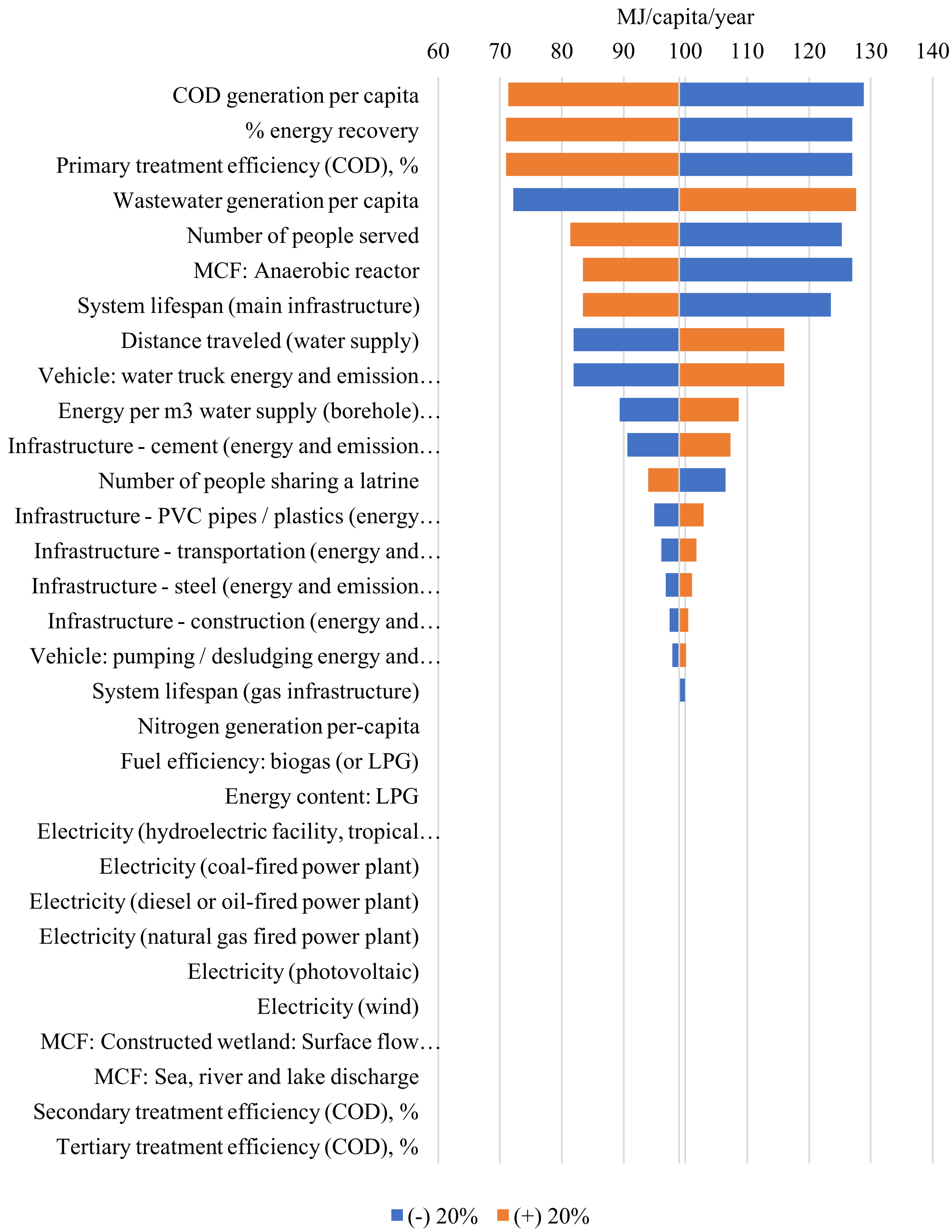


Figure B.9: Sensitivity analysis: Energy use: Decentralized sewered, India

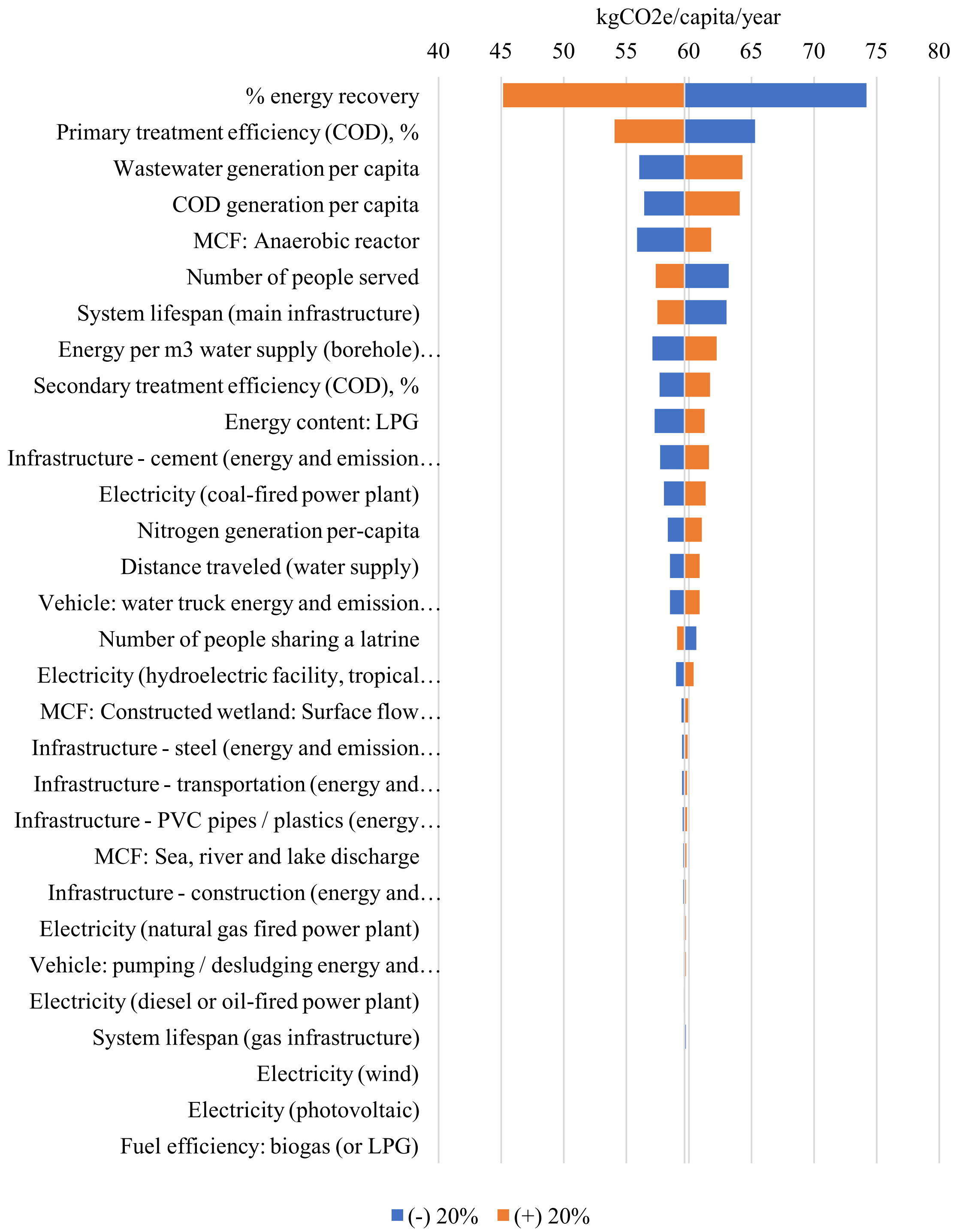


Figure B.10: Sensitivity analysis: Global warming potential (GWP): Decentralized sewered, India

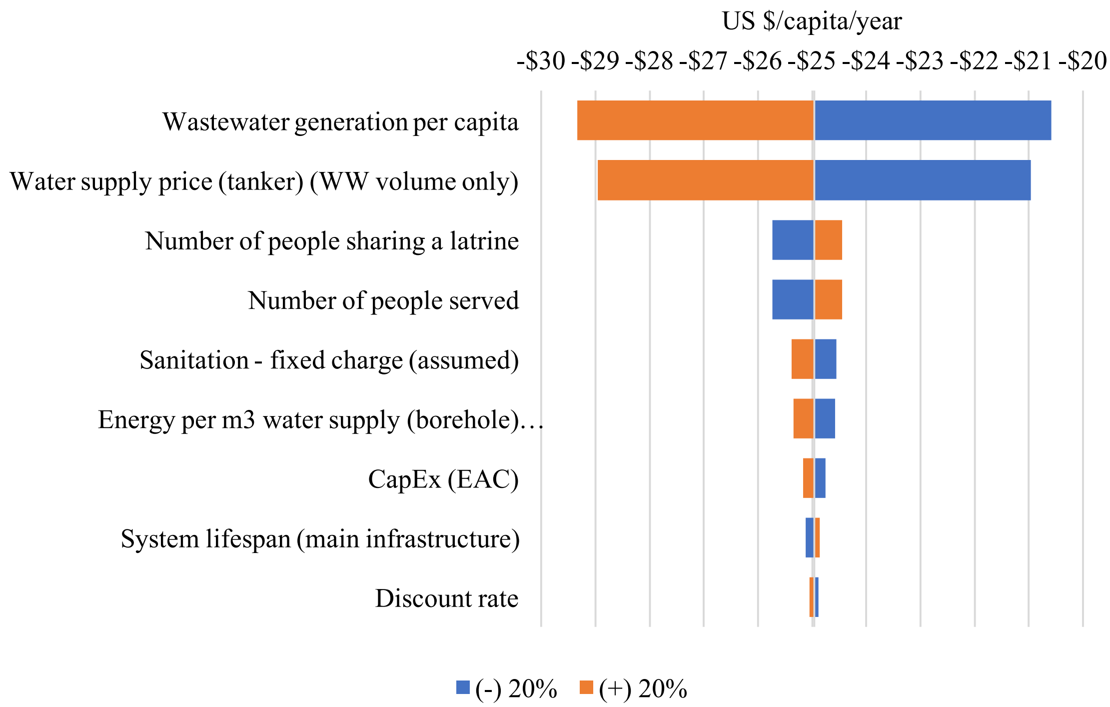


Figure B.11: Sensitivity analysis: Equivalent annual cost (EAC), User: Decentralized sewered, India

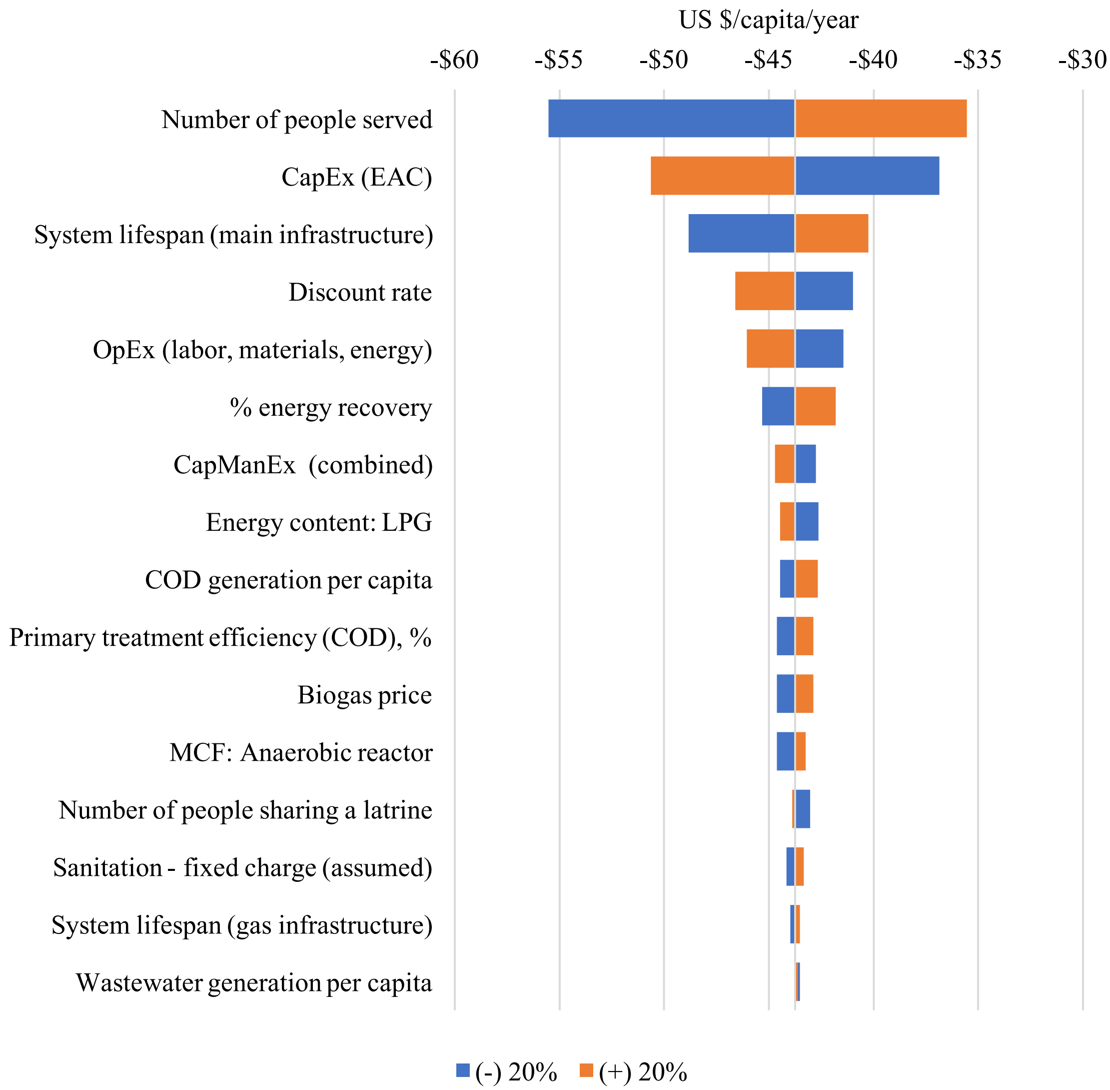


Figure B.12: Sensitivity analysis: Equivalent annual cost (EAC), Agency: Decentralized sewered, India

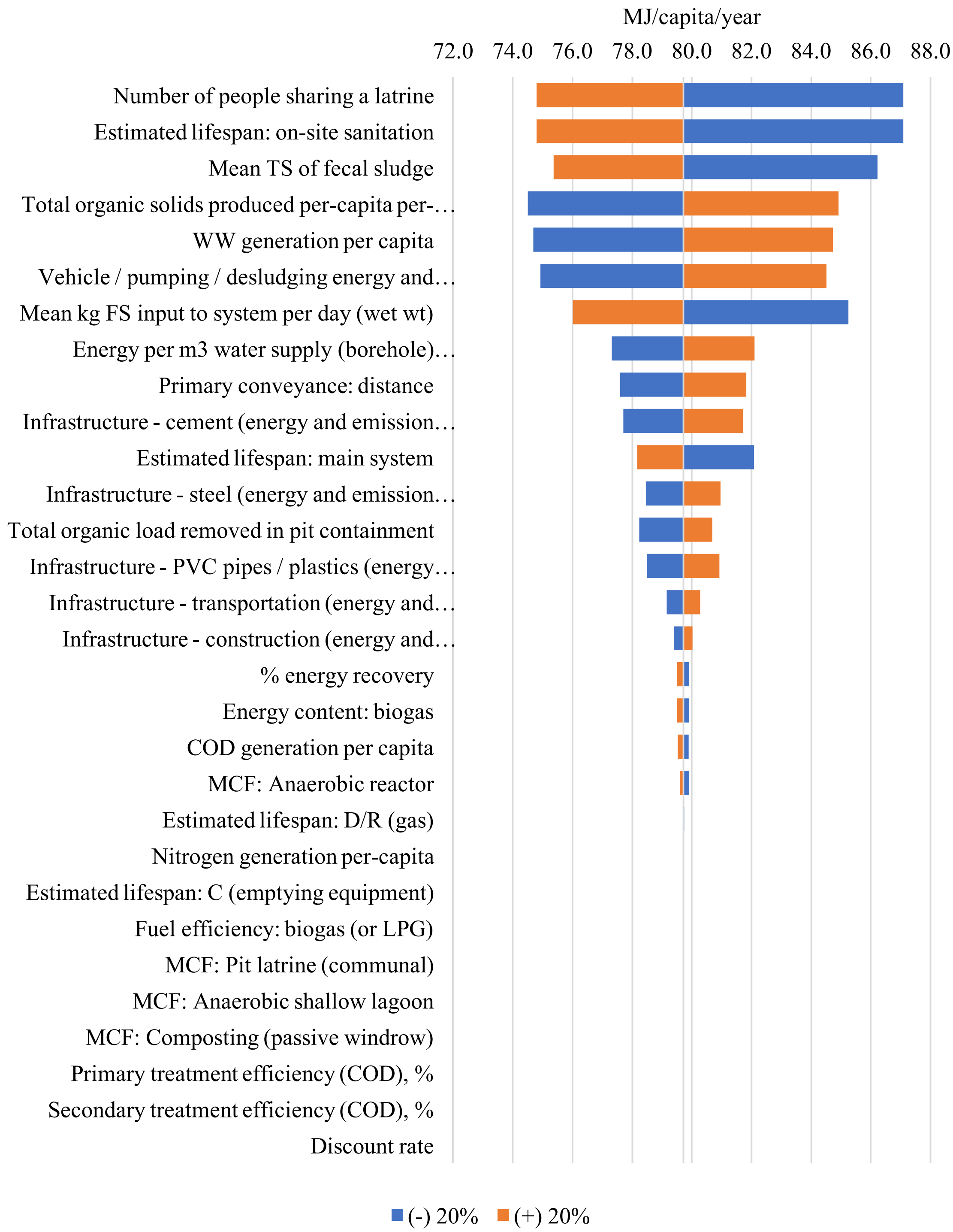


Figure B.13: Sensitivity analysis: Energy use: Non-sewered, India

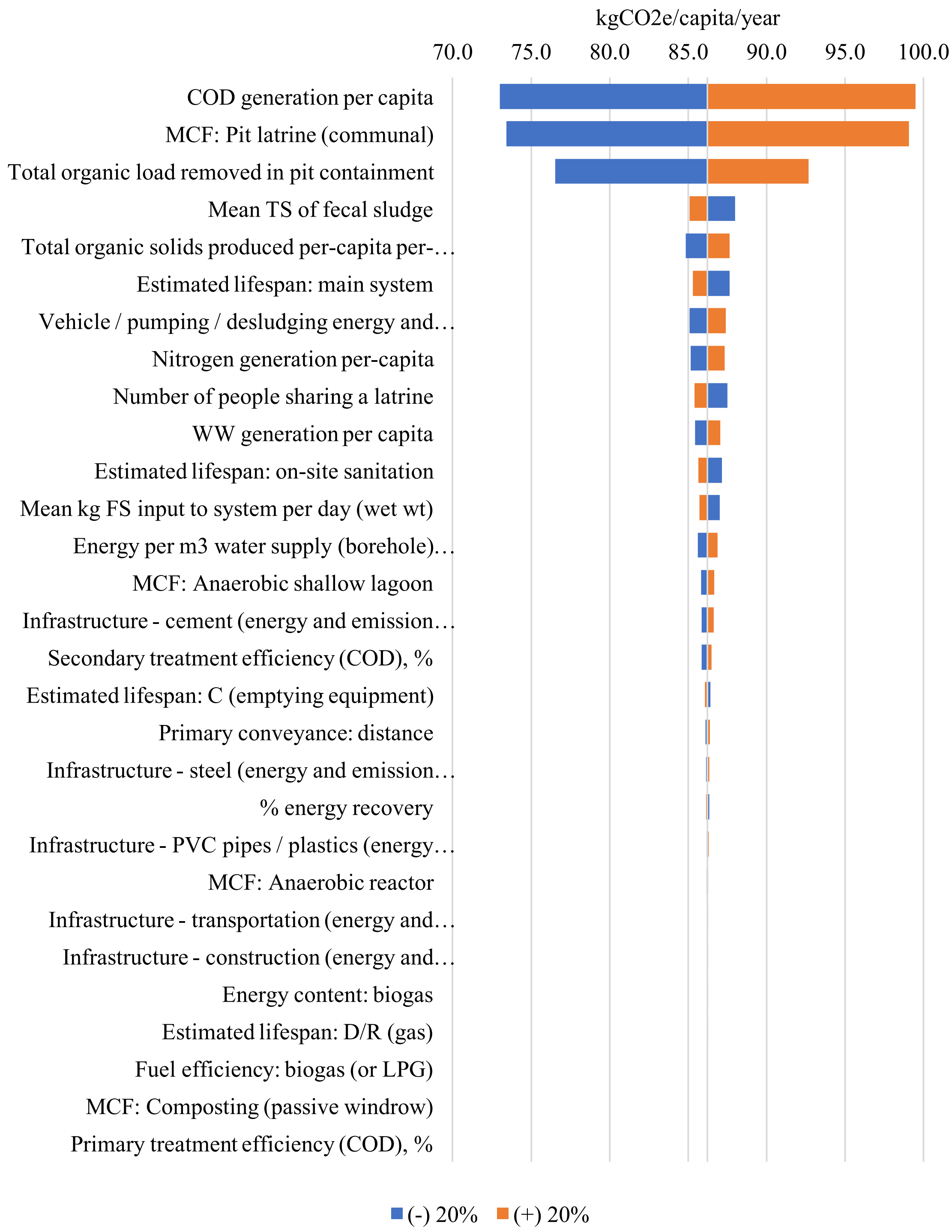


Figure B.14: Sensitivity analysis: Global warming potential (GWP): Non-sewered, India

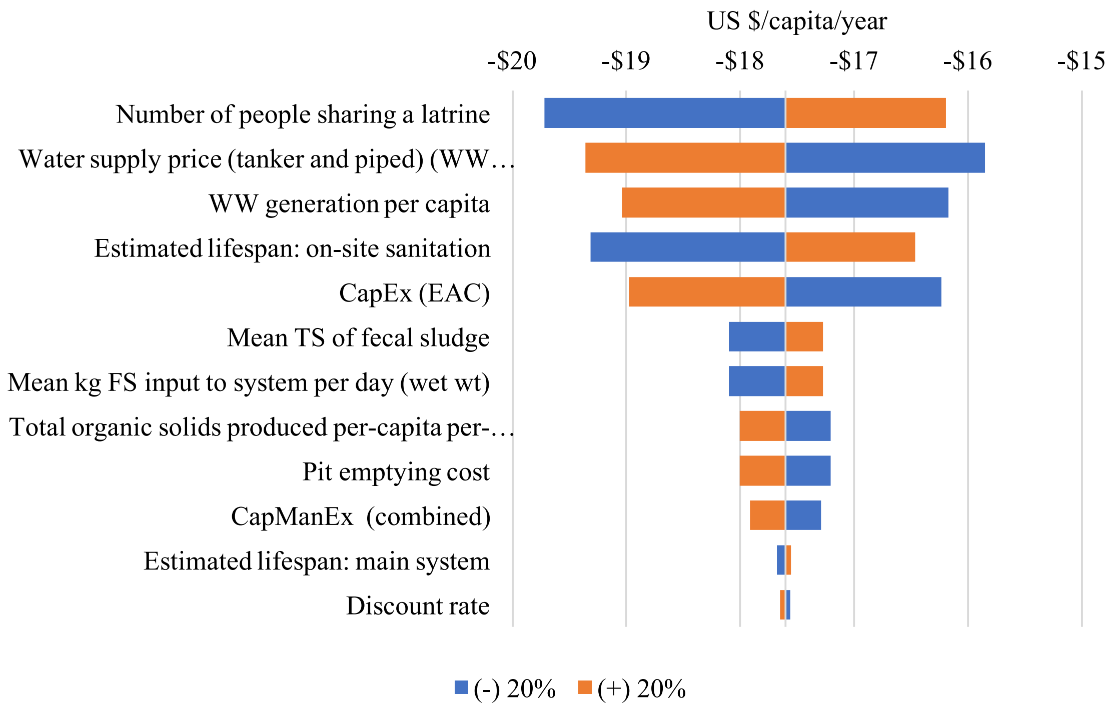


Figure B.15: Sensitivity analysis: Equivalent annual cost (EAC), User: Non-sewered, India

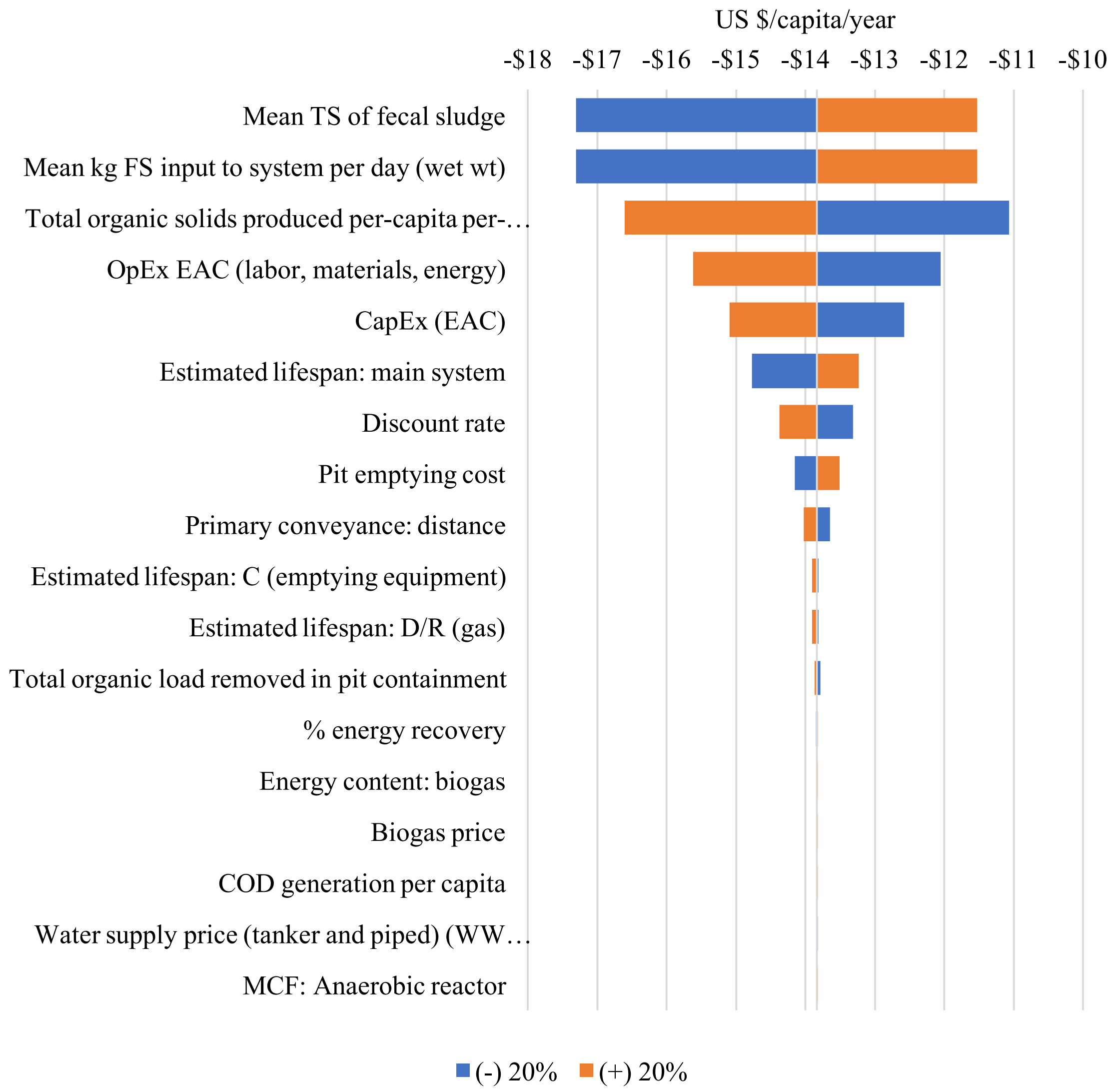


Figure B.16: Sensitivity analysis: Equivalent annual cost (EAC), Agency: Non-sewered, India

B.3 Uncertainty analysis: model variables, baseline values and ranges

Table B.80: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Energy use: Decentralized Sewered, Zambia

Variable	Baseline value	Uncertainty range (worst case)	Uncertainty range (best case)	Units	Baseline reference	Uncertainty reference and notes
COD generation *per capita per* day	62	56	67.4	gCOD/ *capita*/day	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) using data from Laramee et al. (2017). See Appendix B.4).
Primary treatment efficiency (% COD reduction)	61%	58%	64%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Biogas recovery (%)	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Fuel efficiency: biogas (or LPG)	73%	68%	78%	%	Anozie et al. 2007	Assume 7% uncertainty range in efficiency *per* Figure 3 (MacCarty et al., 2010).
Fuel efficiency: charcoal	25%	30%	20%	%	Anozie et al. 2007	Assume 20% uncertainty range in efficiency *per* Figure 3 (MacCarty et al., 2010). Similar variation is reported for mbaula cookstoves in Zambia (Kaoma and Kasali, 1994).
Number of people served	394	374	414	people	Laramee et al. (2017)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
MCF: Anaerobic reactor	0.9	0.8	1.0	-	Table 6.3 (IPCC, 2006)	Table 6.7: +/- 10% uncertainty range recommended for digester (IPCC, 2006).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Conventional fuel: % charcoal use	78%	72%	84%	%	Laramee et al. (2017)	Assume uncertainty equal to standard deviation (SD=5%) (Laramee et al., 2017).
Wastewater generation *per capita*	74	80	69	LPCD	Laramee et al. (2017)	Assume uncertainty equal to 95% confidence interval (=6 LPCD) (Laramee et al., 2017). See Appendix B.4).

Table B.81: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Global warming potential (GWP): Decentralized Sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Biogas recovery (%)	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Primary treatment efficiency (% COD reduction)	61%	58%	64%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Fuel efficiency: biogas (or LPG)	73%	68%	78%	%	Anozie et al. 2007	Assume 7% uncertainty range in efficiency *per* Figure 3 (MacCarty et al., 2010).
Fuel efficiency: charcoal	25%	30%	20%	%	Anozie et al. 2007	Assume 20% uncertainty range in efficiency *per* Figure 3 (MacCarty et al., 2010). Similar variation is reported for mbaula cookstoves in Zambia (Kaoma and Kasali, 1994).
COD generation *per capita per* day [a]	62	56	67.4	gCOD/ *capita*/day	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) using data from Laramee et al. (2017). See Appendix B.4).
Wastewater generation *per capita*	74	80	69	LPCD	Laramee et al. (2017)	Assume uncertainty equal to 95% confidence interval (=6 LPCD) (Laramee et al., 2017). See Appendix B.4).
Conventional fuel: % charcoal use	78%	72%	84%	%	Laramee et al. (2017)	Assume uncertainty equal to standard deviation (SD=5%) (Laramee et al.. 2017).
Number of people served	394	374	414	people	Laramee et al. (2017)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012).
MCF: Anaerobic reactor [a]	0.9	0.8	1.0	-	Table 6.3 (IPCC, 2006)	Table 6.7: +/- 10% uncertainty range recommended for digester (IPCC, 2006).

[a] Worst case and best case uncertainty range flips at approximately 70% and 50% energy recovery, respectively.

Table B.82: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), User: Decentralized Sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Wastewater generation *per capita*	74	80	69	LPCD	Laramee et al. (2017)	Assume uncertainty equal to 95% confidence interval (=6 LPCD) (Laramee et al., 2017). See Appendix B.4).
Water supply tariff: 1st block (0 – 6m3/month)	2.66	3.03	2	ZMW/ m3	1st block tariff - NWWSSC (NWASCO, 2015)	Low to high range of 1st block tariffs used by water and sewerage utilities in Zambia (NWASCO, 2015).
Water supply tariff: 2nd block (6 – 20m3/month)	4.13	4.37	2.45	ZMW/ m3	2nd block tariff - NWWSSC (NWASCO, 2015)	Low to high range of 2nd block tariffs used by water and sewerage utilities in Zambia (NWASCO, 2015).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating) (Newnan et al., 2012).
Sewerage surcharge	40%	40%	20%	%	Surcharge for NWWSSC (NWASCO, 2015)	Low to high range of sewerage surcharge for water and sewerage utilities in Zambia (NWASCO, 2015).
Number of people served	394	374	414	people	Laramee et al. (2017)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Capital expenditure (CapEx) EAC - User	($5)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	BORDA (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).

Table B.83: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), Agency: Decentralized Sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Number of people served	394	374	414	people	Laramee et al. (2017)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Capital expenditure (CapEx) EAC - Agency	($116)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	BORDA (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating) (Newnan et al., 2012).
Discount rate	5%	8%	3%	%	World Bank (2013)	Hutton and Vargues (2016)
Operational expenditures (OpEx) EAC	($69)	+20% of OpEx EAC	-15% of OpEx EAC	US $/ cap/year	BORDA (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Wastewater generation *per capita*	74	80	69	LPCD	Laramee et al. (2017)	95% confidence interval (Laramee et al., 2017). See Appendix B.4).
Water supply tariff: 1st block (0 – 6m3/month)	2.66	3.03	2	ZMW/ m3	1st block tariff - NWWSSC (NWASCO, 2015)	Low to high range of 1st block tariffs used by water and sewerage utilities in Zambia (NWASCO, 2015).
Water supply tariff: 2nd block (6 – 20m3/month)	4.13	4.37	2.45	ZMW/ m3	2nd block tariff - NWWSSC (NWASCO, 2015)	Low to high range of 2nd block tariffs used by water and sewerage utilities in Zambia (NWASCO, 2015).
Sewerage surcharge	40%	40%	20%	%	Surcharge for NWWSSC (NWASCO, 2015)	Low to high range of sewerage surcharge for water and sewerage utilities in Zambia (NWASCO, 2015).
Biogas recovery (%)	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).

Table B.84: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Energy use: Non-sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Total organic load removed in pit containment	88%	91%	85%	%	Computed as a function of COD content at varying pit depths: Figure 1 - Nwaneri et al. (2008)	Uncertainty range computed based on error bars shown in Figure 1 - Nwaneri et al. (2008)
On-site sanitation lifespan	14	11.2	16.1	years	Average of various literature values: Hutton and Vargues (2016): 8yrs; WRC (2007): 10yrs; Tilley et al. (2014): 20yrs.	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Number of people sharing a latrine [a]	5.0	4.5	5.4	people / latrine	Laramee et al. (2017)	Uncertainty range: 95% confidence interval = +/- 0.46. (Laramee et al., 2017).
Biogas recovery (%) [b]	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Primary treatment efficiency (% COD reduction)	22%	19%	26%	%	COD removed computed based on mean biogas production (see methods) WASAZA/ BORDA (2013).	Uncertainty range for COD removal computed based on minimum and maximum biogas production measured (see methods) WASAZA/BORDA (2013).
COD generation *per capita per* day [c]	62	56	67.4	gCOD/ *capita*/ day	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) using data from Laramee et al. (2017). See Appendix B.4).
Cement – energy factor (infrastructure)	6.0	6.0	5.2	MJ/kg cement	SimaPro: Cement, Portland {RoW} | market for | Alloc Def, S	Simapro: Maximum energy factor: Cement, Portland {RoW} | market for | Alloc Def, S; Minimum energy factor: Cement, Portland {US} | market for | Alloc Def, S
MCF: Anaerobic	0.9	0.8	1.0	-	Table 6.3 (IPCC, 2006)	Table 6.7: +/- 10% uncertainty range

reactor						recommended for digester (IPCC, 2006).
Mean total solids (TS, %) of fecal sludge	17%	11%	22%	%	WASAZA/ BORDA (2013)	95% confidence interval (+/-5.5%). Shapiro-Wilk Test for normality: W=0.961, p-value=0.840 (WASAZA/BORDA, 2013)
Mean kg of fecal sludge input *per* day (wet weight)	912.0	799.0	1025.0	kg/day	WASAZA/ BORDA (2013)	95% confidence interval (WASAZA/ BORDA, 2013). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)

[a] Mean number of people *per* latrine measured for non-sewered Zambia system is 17.7 (SD=8.6) (WSUP, 2013). However, to maintain an equivalent basis for comparison between systems, the number of people *per* latrine is assumed to be equal to figure for decentralized sewered system in Zambia.
[b] Equivalent % biogas energy recovery assumed for baseline and uncertainty ranges to compare systems on a standard basis (% recovery based on Laramee et al. 2017).
[c] Equivalent COD generation *per capita per* day assumed for baseline and uncertainty ranges for Zambia systems to compare on a standard basis (*per capita* COD based on Laramee et al., 2017).

Table B.85: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Global warming potential (GWP): Non-sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
MCF: Pit latrine	0.5	0.25	0.75	-	Table 6.3 – assume 'Dry climate, ground water table lower than latrine, communal (many users)' (IPCC, 2006).	Table 6.7: +/- 50% uncertainty range recommended for latrines (IPCC, 2006).
Total organic load removed in pit containment	88%	91%	85%	%	Computed as a function of COD content at varying pit depths: Figure 1 - Nwaneri et al. (2008)	Uncertainty range computed based on error bars shown in Figure 1 - Nwaneri et al. (2008).
COD generation *per capita per* day [a]	62	56	67.4	gCOD/ *capita*/ day	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) using data from Laramee et al. (2017). See Appendix B.4).

[a] Equivalent COD generation *per capita per* day assumed for baseline and uncertainty ranges for Zambia systems to compare on a standard basis (*per capita* COD based on Laramee et al., 2017).

Table B.86: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), User: Non-sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Number of people sharing a latrine [a]	5.0	4.5	5.4	people / latrine	Laramee et al. (2017)	Uncertainty range: 95% confidence interval = +/- 0.46. (Laramee et al., 2017).
On-site sanitation lifespan	14	11.2	16.1	years	Average of various literature values: Hutton and Vargues (2016): 8yrs; WRC (2007): 10yrs; Tilley et al. (2014): 20yrs.	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Capital expenditure (CapEx) EAC - User	($8)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ *capita*/ year	BORDA (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Mean total solids (TS, %) of fecal sludge	17%	11%	22%	%	WASAZA/ BORDA (2013)	95% confidence interval (+/-5.5%). Shapiro-Wilk Test for normality: W=0.961, p-value=0.840 (WASAZA/BORDA, 2013)
Mean kg of fecal sludge input *per* day (wet weight)	912	799	1025	kg/day	WASAZA/ BORDA (2013)	95% confidence interval (WASAZA/ BORDA, 2013). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)
Pit emptying cost	14.9	17.9	12.7	ZMW/ *capita*/ year	Calculated based on number of barrels emptied, cost and estimated number of people served (WSUP, 2016).	Assume range from baseline: 20% greater for worst case; 15% lower for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)

[a] Mean number of people *per* latrine measured for non-sewered Zambia system is 17.7 (SD=8.6) (WSUP, 2013). However, to maintain an equivalent basis for comparison between systems, the number of people *per* latrine is assumed to be equal to figure for decentralized sewered system in Zambia.

Table B.87: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), Agency: Non-sewered, Zambia

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Operational expenditure (OpEx) EAC - Agency	($9)	+20% of OpEx EAC	-15% of OpEx EAC	US $/ *capita*/ year	WSUP (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Mean total solids (TS, %) of fecal sludge	17%	11%	22%	%	WASAZA/ BORDA (2013)	95% confidence interval (+/-5.5%). Shapiro-Wilk Test for normality: W=0.961, p-value=0.840 (WASAZA/BORDA, 2013)
Mean kg of fecal sludge input *per* day (wet weight)	912	799	1025	kg/day	WASAZA/ BORDA (2013)	95% confidence interval (WASAZA/ BORDA, 2013). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)
Pit emptying cost	14.9	12.7	17.9	ZMW/ *capita*/ year	Calculated based on number of barrels emptied, cost and estimated number of people served (WSUP, 2016).	Assume range from baseline: 15% greater for worst case; 20% lower for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating) (Newnan et al., 2012).
Capital expenditure (CapEx) EAC - Agency	($2)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ *capita*/ year	BORDA (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Total organic load removed in pit containment	88%	91%	85%	%	Computed as a function of COD content at varying pit depths: Figure 1 - Nwaneri et al. (2008)	Uncertainty range computed based on error bars shown in Figure 1 - Nwaneri et al. (2008).
Discount rate	5%	8%	3%	%	World Bank (2013)	Hutton and Vargues (2016)

Table B.88: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Energy use: Decentralized Sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
COD generation *per capita per* day	54	52	55	gCOD/ *capita*/day	Reynaud and Buckley (2015); Miller (2011)	95% confidence interval computed with data from Reynaud and Buckley (2015) and Miller (2011)
Biogas recovery (%) [a]	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Primary treatment efficiency (% COD reduction)	73%	68%	77%	%	Reynaud and Buckley (2015)	95% confidence interval computed with data from Reynaud and Buckley (2015)
Wastewater generation *per capita*	29	27	30	LPCD	Reynaud and Buckley, 2015	95% confidence interval computed with data from Reynaud and Buckley (2015)
Number of people served	575	546	604	people	Reynaud and Buckley, 2015	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
MCF: Anaerobic reactor	0.9	0.8	1.0	-	Table 6.3 (IPCC, 2006)	Table 6.7: +/- 10% uncertainty range recommended for digester (IPCC, 2006).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Distance traveled: Water supply *via* tanker truck	4.0	4.8	3.4	km	CDD, 2016	Assume range from baseline: 20% greater for worst case; 15% lower for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Vehicle energy factor (water supply tanker truck)	3.66	4.29	2.86	MJ/tkm	SimaPro: "Transport, truck 10-20t, EURO5, 80%LF, empty return/GLO Energy"	Uncertainty: 8 - 12L/100km (diesel = 35.8MJ/L) (estimate from CDD, 2016)

[a] Equivalent % biogas energy recovery assumed for baseline and uncertainty ranges to compare systems on a standard basis (% recovery based on Laramee et al. 2017).

Table B.89: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Global warming potential (GWP): Decentralized Sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Biogas recovery (%) [a]	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).
Primary treatment efficiency (% COD reduction)	73%	68%	77%	%	Reynaud and Buckley (2015)	95% confidence interval computed with data from Reynaud and Buckley (2015)
Wastewater generation *per capita*	29	27	30	LPCD	Reynaud and Buckley, 2015	95% confidence interval computed with data from Reynaud and Buckley (2015)
COD generation *per capita per* day	54	52	55	gCOD/ *capita*/day	Reynaud and Buckley (2015); Miller (2011)	95% confidence interval computed with data from Reynaud and Buckley (2015) and Miller (2011)
MCF: Anaerobic reactor	0.9	0.8	1.0	-	Table 6.3 (IPCC, 2006)	Table 6.7: +/- 10% uncertainty range recommended for digester (IPCC, 2006).
Number of people served	575	546	604	people	Reynaud and Buckley, 2015	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Energy *per* m3 water supply (borehole) (kWh/m3)	1.3	1.7	0.9	kWh/ m3	Computed based on borehole depth 225m (personal communication: CDD, 2016)	Computed based on borehole depth ranging from 150 - 300m (personal communication: CDD, 2016)

[a] Equivalent % biogas energy recovery assumed for baseline and uncertainty ranges to compare systems on a standard basis (% recovery based on Laramee et al. 2017).

Table B.90: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), User - Decentralized Sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Wastewater generation *per capita*	29	27	30	LPCD	Reynaud and Buckley, 2015	95% confidence interval computed with data from Reynaud and Buckley (2015)
Water supply price (tanker truck)	58	70	50	INR/ m3	Personal communication: CDD (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Number of people sharing a latrine	4.8	4.6	5.0	people/ latrine	Reynaud and Buckley, 2015	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Number of people served	575	546	604	people	Reynaud and Buckley, 2015	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Sanitation - fixed charge (assumed)	14.0	16.8	11.9	INR/ HH	Sanitation fixed charge - utilities in Bangalore (IBNET, 2016).	Assume range from baseline: 20% greater for worst case; 15% lower for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Energy *per* m3 water supply (borehole) (kWh/m3)	1.3	1.7	0.9	kWh/ m3	Computed based on borehole depth 225m - personal communication: CDD (2016)	Computed based on borehole depth ranging from 150 - 300m (personal communication: CDD, 2016)
Capital expenditure (CapEx) EAC - User	($1)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)

Table B.91: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), Agency - Decentralized Sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Number of people served	575	546	604	people	Reynaud and Buckley, 2015	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Capital expenditure (CapEx) EAC - Agency	($34)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Discount rate	5%	8%	3%	%	World Bank (2013)	Hutton and Vargues (2016)
Operational expenditure (OpEx) EAC - Agency	($12)	+20% of OpEx EAC	-15% of OpEx EAC	US $/ *capita*/ year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Biogas recovery (%) [a]	82%	76%	88%	%	Laramee et al. (2017)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from Laramee et al. (2017). See Appendix B.4).

[a] Equivalent % biogas energy recovery assumed for baseline and uncertainty ranges to compare systems on a standard basis (% recovery based on Laramee et al. 2017).

Table B.92: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Energy use: Non-sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Number of people sharing a latrine	4.4	4.2	4.6	people/ latrine	CDD (2015)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
On-site sanitation lifespan	14	11.2	16.1	years	Average of various literature values: Hutton and Vargues (2016): 8yrs; WRC (2007): 10yrs; Tilley et al. (2014): 20yrs.	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Mean total solids (TS, %) of fecal sludge	4.2%	3.2%	5.3%	%	CDD (2016) (FSTP_Devanahalli_ Dashboard)	95% confidence interval (CDD, 2016). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)
Wastewater generation *per capita*	7	10	4	LPCD	CDD (2015); Mara (1981)	Range of values for pour-flush toilets)Mara (1981)
Vehicle energy factors: water supply, sludge transport, pumping, desludging	Varies	+20%	-15%	MJ/tkm	SimaPro (various)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Mean kg of fecal sludge input *per* day (wet weight)	1978	1232	2724	kg/day	CDD (2016) (FSTP_Devanahalli_ Dashboard)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from CDD (2016). See Appendix B.4).
Energy *per* m3 water supply (borehole) (kWh/m3)	1.3	1.7	0.9	kWh/ m3	Computed based on borehole depth 225m (personal communication: CDD, 2016)	Computed based on borehole depth ranging from 150 - 300m (personal communication: CDD, 2016)
Primary conveyance: distance	10.5	14.3	6.7	km/ round-trip	CDD (2016) (FSTP_Devanahalli_D ashboard)	95% confidence interval (CDD, 2016).

Cement – energy factor (infrastructure)	6.0	6.0	5.2	MJ/kg cement	SimaPro: Cement, Portland {RoW} | market for | Alloc Def, S	Simapro: Maximum energy factor: Cement, Portland {RoW} | market for | Alloc Def, S; Minimum energy factor: Cement, Portland {US} | market for | Alloc Def, S
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)

Table B.93: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Global warming potential (GWP): Non-sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
COD generation *per capita per* day [a]	54	55	52	gCOD/ *capita*/ day	Reynaud and Buckley (2015); Miller (2011)	95% confidence interval computed with data from Reynaud and Buckley (2015) and Miller (2011)
MCF: Pit latrine	0.7	1.0	0.35	-	Table 6.3 – assume 'Wet climate/flush water use, ground water table higher than latrine' (IPCC, 2006).	Table 6.7: +/- 50% uncertainty range recommended for latrines (IPCC, 2006).
Total organic load removed in pit containment	88%	91%	85%	%	Computed as a function of COD content at varying pit depths: Figure 1 - Nwaneri et al. (2008)	Uncertainty range computed based on error bars shown in Figure 1 - Nwaneri et al. (2008).

[a] Equivalent COD generation *per capita per* day assumed for baseline and uncertainty ranges for India systems to compare on a standard basis (*per capita* COD based on Reynaud and Buckley, 2015).

Table B.94: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), User: Non-sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Number of people sharing a latrine	4.4	4.2	4.6	people/ latrine	CDD (2015)	Table 6.7: +/- 5% uncertainty range recommended for population (IPCC, 2006).
Water supply tariff (tanker)	133	167	100	INR/m3	CDD (2016)	CDD (2016)
Water supply price (municipal piped, fixed charge)	100	120	80	INR/ month	CDD (2016)	CDD (2016)
Wastewater generation *per capita*	7	10	4	LPCD	CDD (2015); Mara (1981)	Range of values for pour-flush toilets (Mara, 1981)
On-site sanitation lifespan	14	11.2	16.1	years	Average of various literature values: Hutton and Vargues (2016): 8yrs; WRC (2007): 10yrs; Tilley et al. (2014): 20yrs.	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Capital expenditure (CapEx) EAC - User	($5)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Mean total solids (TS, %) of fecal sludge	4.2%	3.2%	5.3%	%	CDD (2016) (FSTP_Devanahalli_ Dashboard)	95% confidence interval (CDD, 2016). See Appendix B.4).
Mean kg of fecal sludge input *per* day (wet weight)	1978	1232	2724	kg/day	CDD (2016) (FSTP_Devanahalli_ Dashboard)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from CDD (2016). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)
Pit emptying fee	34	41	29	INR/ *capita*/ year	CDD (2016) (FSTP_Devanahalli_ Dashboard)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).

Table B.95: Uncertainty analysis: variables with greatest effect on model output (in order of importance), baseline values and uncertainty range – Equivalent annual cost (EAC), Agency: Non-sewered, India

Variable	**Baseline value**	**Uncertainty range (worst case)**	**Uncertainty range (best case)**	**Units**	**Baseline reference**	**Uncertainty reference and notes**
Mean total solids (TS, %) of fecal sludge	4.2%	3.2%	5.3%	%	CDD (2016) (FSTP_Devanahalli_ Dashboard)	95% confidence interval (CDD, 2016). See Appendix B.4).
Mean kg of fecal sludge input *per* day (wet weight)	1978	1232	2724	kg/day	CDD (2016) (FSTP_Devanahalli_ Dashboard)	Bootstrap 95% confidence interval for the mean computed (R, 2017) with data from CDD (2016). See Appendix B.4).
Total organic solids produced *per capita per* day (dry weight)	39.0	50.8	27.2	gPCD	Rose et al. (2016)	95% confidence interval calculated using data in Table 3 - Rose et al. (2016)
Operational expenditure (OpEx) EAC - Agency	($9)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
Capital expenditure (CapEx) EAC - Agency	($6)	+20% of CapEx EAC	-15% of CapEx EAC	US $/ cap/year	CDD (2016)	Assume range from baseline: 20% greater for worst case; 15% lower for best case (Newnan et al., 2012).
System lifespan (main infrastructure)	20	16	23	years	BORDA – personal communication (2016)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (similar rationale to enginering cost estimating). (Newnan et al., 2012)
Discount rate	5%	8%	3%	%	World Bank (2013)	Hutton and Vargues (2016)
Pit emptying revenue [a]	27	33	23	INR/ *capita*/ year	CDD (2016) (FSTP_Devanahalli_ Dashboard)	Assume range from baseline: 20% lower for worst case; 15% greater for best case (Newnan et al., 2012).

[a] Pit emptying revenue (agency) *per capita* is computed as less than pit emptying fee (user) *per capita* as users also pay private operators for pit emptying services.

B.4 Normality tests and figures (QQ Plot, Histogram)

COD generation per capita per day (gCOD/cap/day): Decentralized sewered (DS), Zambia

Raw data:

Obs.	gCOD production *per capita per* day
1	33
2	42
3	42
4	46
5	46
6	52
7	54
8	60
9	61
10	63
11	63
12	66
13	67
14	67
15	68
16	69
17	69
18	69
19	69
20	71
21	72
22	74
23	77
24	78
Mean	62
St Dev	12
Count	24
Skew	-0.9

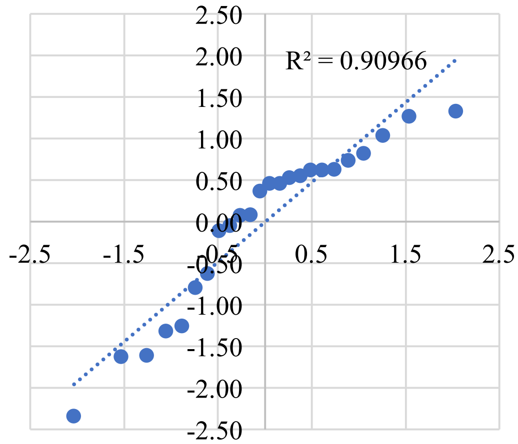
Q-Q plot:

Cumulative probability	Z-score	Standardize
0.02	-2.04	-2.33
0.06	-1.53	-1.62
0.10	-1.26	-1.60
0.15	-1.05	-1.31
0.19	-0.89	-1.25
0.23	-0.74	-0.79
0.27	-0.61	-0.62
0.31	-0.49	-0.10
0.35	-0.37	-0.04
0.40	-0.26	0.08
0.44	-0.16	0.09
0.48	-0.05	0.37
0.52	0.05	0.46
0.56	0.16	0.47
0.60	0.26	0.54
0.65	0.37	0.56
0.69	0.49	0.62
0.73	0.61	0.63
0.77	0.74	0.63
0.81	0.89	0.74
0.85	1.05	0.82
0.90	1.26	1.04
0.94	1.53	1.27
0.98	2.04	1.33

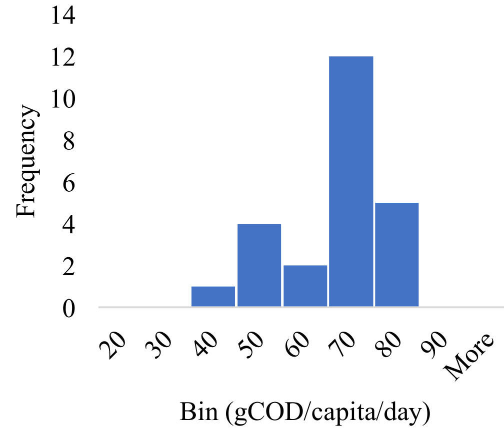
Histogram

Bin	Frequency
20	0
30	0
40	1
50	4
60	2
70	12
80	5
90	0
More	0

Q-Q Plot: COD per capita per day (DS, Zambia)



Histogram: gCOD per capita per day (DS, Zambia)



Shapiro-Wilk test for normality:

W = 0.90465, P-value=0.02704

p-value < 0.1, therefore reject null hypothesis - conclude data is not normally distributed

Computed a bootstrap confidence interval for the mean (computed in R statistical software)

Shapiro-Wilk test for normality of bootstrap means:

W=0.991; p-value=0.278

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume bootstrap means resampled data normally distributed.

Descriptive Statistics (of bootstrap means):

Mean	61	gCOD/*capita*/day
Median	62	gCOD/*capita*/day
St Dev	3	gCOD/*capita*/day
95% Conf. Interval (2.5%)	56	gCOD/*capita*/day
95% Conf. Interval (97.5%)	67	gCOD/*capita*/day

Primary treatment efficiency (% COD reduction): Decentralized sewered (DS), Zambia

Raw data:

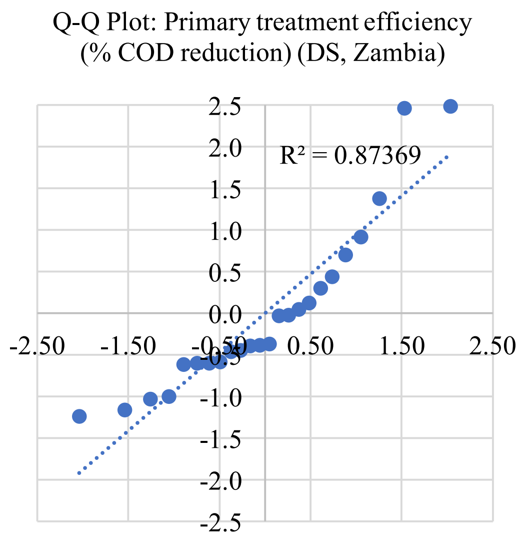
Obs.	gCOD production *per capita per* day
1	53%
2	54%
3	55%
4	55%
5	57%
6	57%
7	57%
8	57%
9	58%
10	58%
11	58%
12	58%
13	59%
14	61%
15	61%
16	61%
17	61%
18	62%
19	63%
20	65%
21	66%
22	69%
23	75%
24	75%
Mean	61%
St Dev	6%
Count	24
Skew	1.3

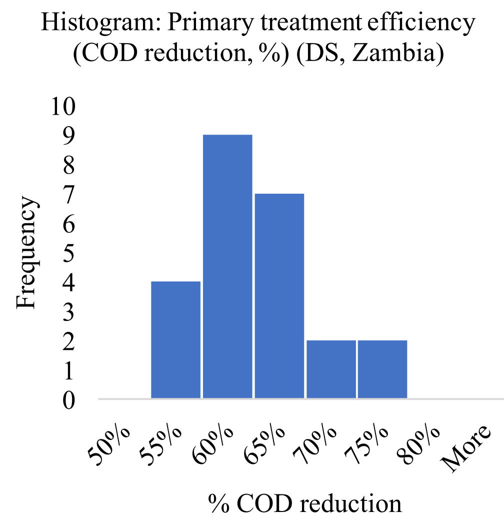
Q-Q plot:

Cumulative probability	Z-score	Standardize
0.02	-2.04	-1.24
0.06	-1.53	-1.15
0.10	-1.26	-1.03
0.15	-1.05	-1.00
0.19	-0.89	-0.61
0.23	-0.74	-0.60
0.27	-0.61	-0.60
0.31	-0.49	-0.58
0.35	-0.37	-0.46
0.40	-0.26	-0.44
0.44	-0.16	-0.39
0.48	-0.05	-0.38
0.52	0.05	-0.36
0.56	0.16	-0.03
0.60	0.26	-0.02
0.65	0.37	0.05
0.69	0.49	0.12
0.73	0.61	0.30
0.77	0.74	0.45
0.81	0.89	0.70
0.85	1.05	0.92
0.90	1.26	1.38
0.94	1.53	2.46
0.98	2.04	2.49

Histogram

Bin	Frequency
50%	0
55%	4
60%	9
65%	7
70%	2
75%	2
80%	0
More	0





Shapiro-Wilk test for normality

W = 0.871, p-value=0.0055

p-value < 0.1, therefore reject null hypothesis - conclude data is not normally distributed

Computed a bootstrap confidence interval for the mean (computed in R statistical software)

Shapiro-Wilk test for normality of bootstrap means:

W=0.991; p-value=0.278

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume bootstrap means resampled data normally distributed.

Descriptive statistics (of bootstrap means):

Mean	61%	%
Median	61%	%
St Dev	2%	%
95% Conf. Interval (2.5%)	58%	%
95% Conf. Interval (97.5%)	64%	%

Test for normality: Biogas recovery (% recovery of total biogas production): Decentralized sewered (DS), Zambia

Raw data:

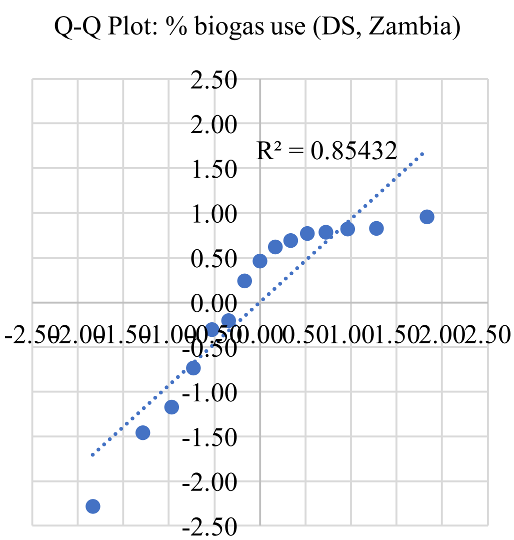
System	Biogas use
1	48%
2	61%
3	65%
4	71%
5	77%
6	79%
7	85%
8	89%
9	91%
10	92%
11	93%
12	93%
13	94%
14	94%
15	96%
Mean	82%
St Dev	15%
Count	15
Skew	-1.1

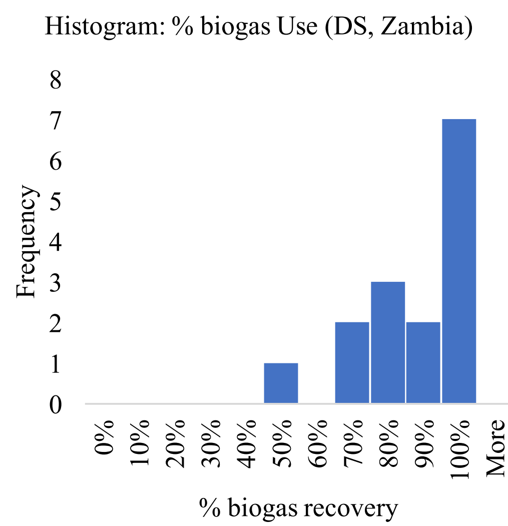
Q-Q plot:

Cumulative probability	Z-score	Standardize
0.03	-1.83	-2.29
0.10	-1.28	-1.46
0.17	-0.97	-1.17
0.23	-0.73	-0.74
0.30	-0.52	-0.31
0.37	-0.34	-0.21
0.43	-0.17	0.24
0.50	0.00	0.46
0.57	0.17	0.62
0.63	0.34	0.69
0.70	0.52	0.77
0.77	0.73	0.78
0.83	0.97	0.82
0.90	1.28	0.83
0.97	1.83	0.96

Histogram

Bin	Frequency
0%	0
10%	0
20%	0
30%	0
40%	0
50%	1
60%	0
70%	2
80%	3
90%	2
100%	7
More	0





Shapiro-Wilk test for normality

W = 0.849, P-value=0.01687

p-value < 0.1, therefore reject null hypothesis - conclude data is not normally distributed

Computed a bootstrap confidence interval for the mean (computed in R statistical software)

Shapiro-Wilk test for normality of bootstrap means:

W=0.995; p-value=0.8045

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume bootstrap means resampled data normally distributed.

Descriptive statistics (of bootstrap means):

Mean	82%	%
Median	82%	%
St Dev	4%	%
95% Conf. Interval (2.5%)	76%	%
95% Conf. Interval (97.5%)	88%	%

Test for normality: Wastewater generation per capita per day (LPCD): Decentralized sewered (DS), Zambia

Raw data:

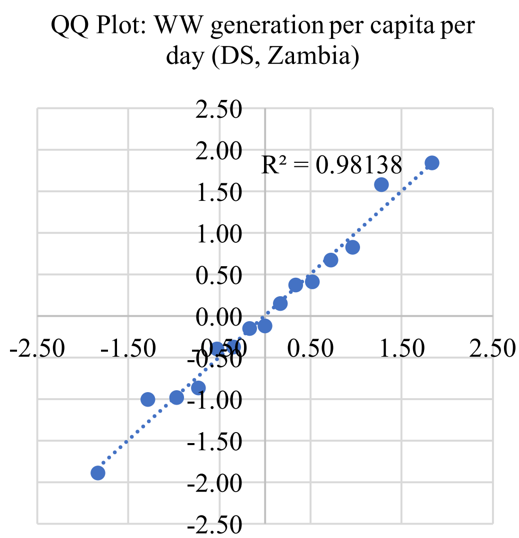
Obs.	**Wastewater generation (LPCD)**
1	55
2	64
3	64
4	65
5	70
6	71
7	73
8	73
9	76
10	78
11	79
12	81
13	83
14	90
15	93
Mean	74
Median	73
St Dev	10
n	15
Skew	0.12
95% Conf. Interval	6

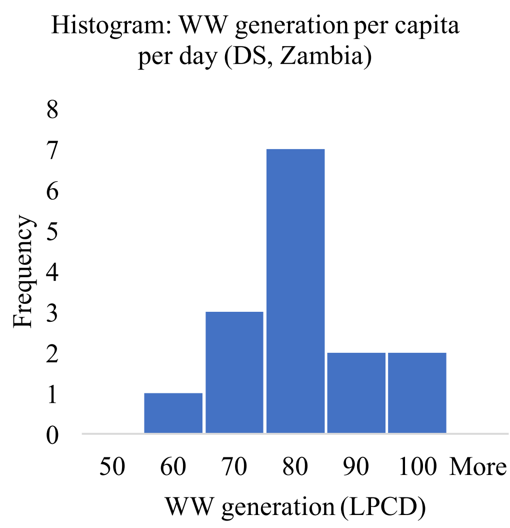
Q-Q plot:

Cumulative probability	**Z-score**	**Standardize**
0.03	-1.83	-1.90
0.10	-1.28	-1.01
0.17	-0.97	-0.99
0.23	-0.73	-0.87
0.30	-0.52	-0.40
0.37	-0.34	-0.38
0.43	-0.17	-0.15
0.50	0.00	-0.13
0.57	0.17	0.14
0.63	0.34	0.37
0.70	0.52	0.40
0.77	0.73	0.67
0.83	0.97	0.82
0.90	1.28	1.58
0.97	1.83	1.84

Histogram

Bin	**Frequency**
50	0
60	1
70	3
80	7
90	2
100	2
More	0





Shapiro-Wilk test for normality

W = 0.981, P-value=0.9767

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume data normally distributed.

Test for normality: Fecal sludge inputs per day (wet weight) – Non-sewered (NS), Zambia

Raw data:

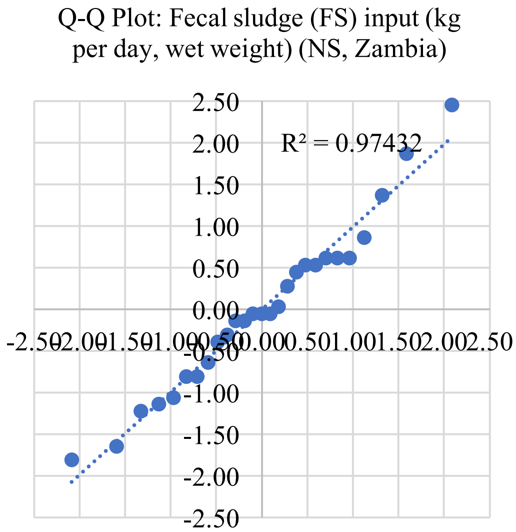
Observations (Weeks)	**Fecal sludge input (LPD)**
1	374
2	423
3	548
4	573
5	598
6	672
7	672
8	722
9	797
10	822
11	872
12	872
13	896
14	896
15	896
16	921
17	996
18	1046
19	1071
20	1071
21	1096
22	1096
23	1096
24	1170
25	1320
26	1469
27	1644
Mean	912
Median	896
St Dev	298
n	27
Skew	0.39
95% Conf. Interval	118

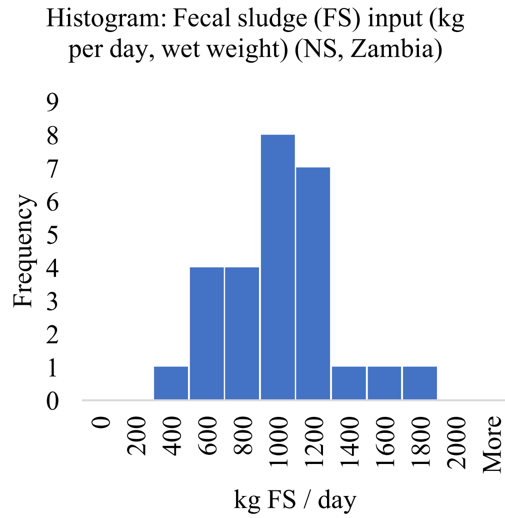
Q-Q plot:

Cumulative probability	**Z-score**	**Standardize**
0.02	-2.09	-1.81
0.06	-1.59	-1.64
0.09	-1.32	-1.22
0.13	-1.13	-1.14
0.17	-0.97	-1.06
0.20	-0.83	-0.81
0.24	-0.70	-0.81
0.28	-0.59	-0.64
0.31	-0.48	-0.39
0.35	-0.38	-0.30
0.39	-0.28	-0.14
0.43	-0.19	-0.14
0.46	-0.09	-0.05
0.50	0.00	-0.05
0.54	0.09	-0.05
0.57	0.19	0.03
0.61	0.28	0.28
0.65	0.38	0.45
0.69	0.48	0.53
0.72	0.59	0.53
0.76	0.70	0.62
0.80	0.83	0.62
0.83	0.97	0.62
0.87	1.13	0.87
0.91	1.32	1.37
0.94	1.59	1.87
0.98	2.09	2.46

Histogram

Bin	**Frequency**
0	0
200	0
400	1
600	4
800	4
1000	8
1200	7
1400	1
1600	1
1800	1
2000	0
More	0





Shapiro-Wilk test for normality

W = 0.97462; p-value = 0.7264

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume data normally distributed.

Test for normality: Fecal sludge (FS) input per day (kg wet weight per day): Non-sewered (NS), India

Raw data: / **Q-Q plot:**

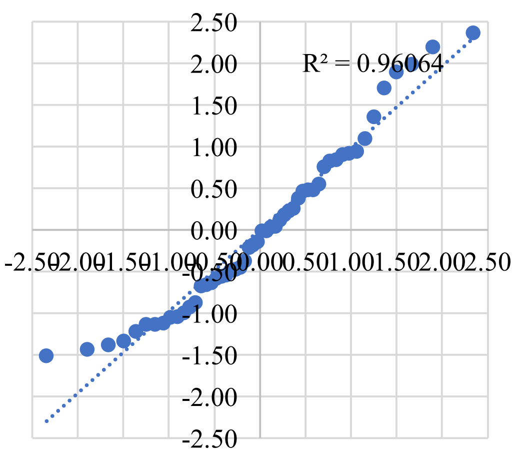
Observations (Week)	**kg fecal sludge *per* day (wet weight)**	**Cumulative probability**	**Z-score**	**Standardize**
1	671	0.01	-2.34	-1.51
2	740	0.03	-1.90	-1.43
3	790	0.05	-1.66	-1.38
4	829	0.07	-1.50	-1.33
5	929	0.09	-1.36	-1.22
6	1000	0.11	-1.25	-1.13
7	1000	0.13	-1.15	-1.13
8	1014	0.14	-1.06	-1.12
9	1075	0.16	-0.98	-1.05
10	1083	0.18	-0.91	-1.04
11	1122	0.20	-0.83	-0.99
12	1180	0.22	-0.77	-0.92
13	1229	0.24	-0.71	-0.87
14	1400	0.26	-0.64	-0.67
15	1414	0.28	-0.59	-0.65
16	1433	0.30	-0.53	-0.63
17	1478	0.32	-0.48	-0.58
18	1500	0.34	-0.42	-0.56
19	1513	0.36	-0.37	-0.54
20	1557	0.38	-0.32	-0.49
21	1571	0.39	-0.27	-0.47
22	1593	0.41	-0.22	-0.45
23	1656	0.43	-0.17	-0.37
24	1800	0.45	-0.12	-0.21
25	1829	0.47	-0.07	-0.18
26	1857	0.49	-0.02	-0.14
27	1971	0.51	0.02	-0.01
28	1971	0.53	0.07	-0.01
29	2013	0.55	0.12	0.04
30	2020	0.57	0.17	0.05
31	2086	0.59	0.22	0.12
32	2136	0.61	0.27	0.18
33	2180	0.63	0.32	0.23
34	2207	0.64	0.37	0.26
35	2314	0.66	0.42	0.39
36	2383	0.68	0.48	0.47
37	2400	0.70	0.53	0.49
38	2400	0.72	0.59	0.49
39	2457	0.74	0.64	0.55
40	2636	0.76	0.71	0.76
41	2700	0.78	0.77	0.83
42	2714	0.80	0.83	0.85
43	2767	0.82	0.91	0.91
44	2780	0.84	0.98	0.92
45	2800	0.86	1.06	0.95
46	2929	0.88	1.15	1.10

Histogram

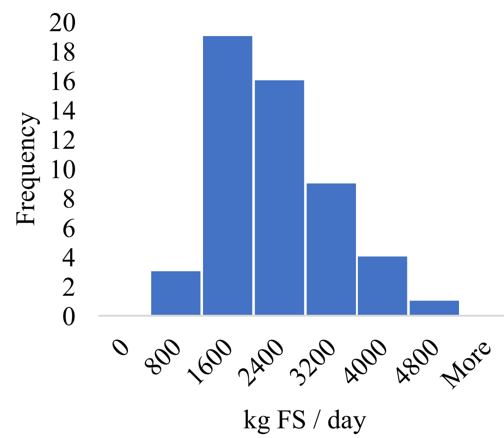
Bin	**Frequency**
0	0
800	3
1600	19
2400	16
3200	9
4000	4
4800	1
More	0

47	3157	0.89	1.25	1.36
48	3457	0.91	1.36	1.71
49	3620	0.93	1.50	1.89
50	3700	0.95	1.66	1.99
51	3886	0.97	1.90	2.20
52	4029	0.99	2.34	2.37
Mean	1980			
St Dev	865			
Count	52			
Skew	0.57			

Q-Q Plot: Fecal sludge (FS) input (kg FS per day, wet weight) (NS, India)



Histogram: Fecal sludge (FS) input (kg FS per day, wet weight) (NS, India)



Shapiro-Wilk test for normality

W = 0.9555, P-value=0.05

p-value < 0.1, therefore reject null hypothesis - conclude data is not normally distributed

Computed a bootstrap confidence interval for the mean (computed in R statistical software)

Shapiro-Wilk test for normality of bootstrap means:

W=0.99054; p-value=0.2141

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume bootstrap means resampled data normally distributed.

Descriptive statistics (of bootstrap means):

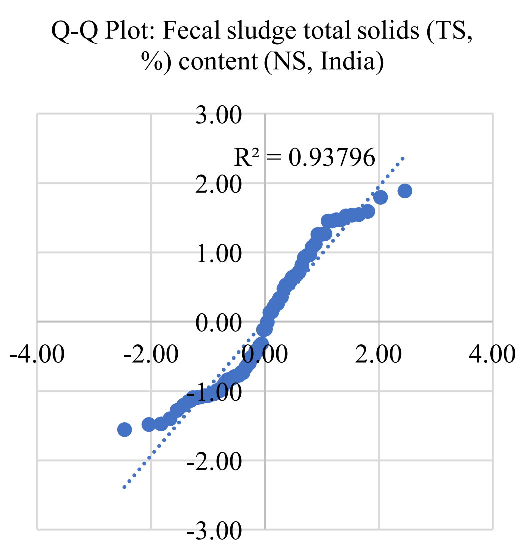
Mean	1974	kg FS / day
Median	1978	kg FS / day
St Dev	209	kg FS / day
95% Conf. Interval (2.5%)	1232	kg FS / day
95% Conf. Interval (97.5%)	2724	kg FS / day

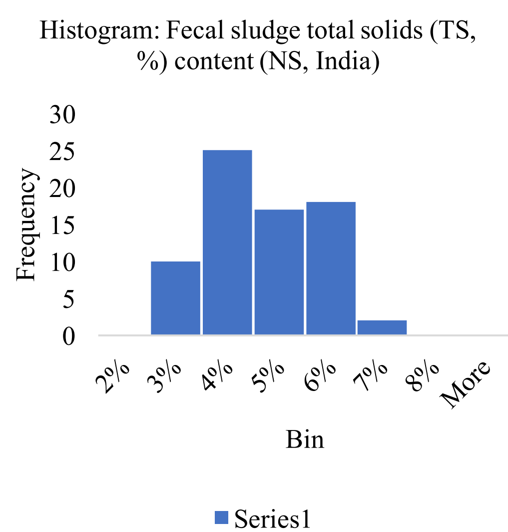
Test for normality: Fecal sludge (FS) total solids content (TS, %): Non-sewered (NS), India

Raw data:		Q-Q plot:		
Observations (Week)	**Total solids content (%)**	**Cumulative probability**	**Z-score**	**Standardize**
1	2.5%	0.01	-2.46	-1.56
2	2.5%	0.02	-2.04	-1.49
3	2.6%	0.03	-1.82	-1.48
4	2.6%	0.05	-1.66	-1.40
5	2.8%	0.06	-1.53	-1.28
6	2.9%	0.08	-1.43	-1.20
7	2.9%	0.09	-1.34	-1.15
8	3.0%	0.10	-1.26	-1.10
9	3.0%	0.12	-1.18	-1.10
10	3.0%	0.13	-1.12	-1.09
11	3.0%	0.15	-1.05	-1.07
12	3.0%	0.16	-1.00	-1.06
13	3.0%	0.17	-0.94	-1.05
14	3.1%	0.19	-0.89	-1.04
15	3.1%	0.20	-0.84	-1.00
16	3.1%	0.22	-0.79	-1.00
17	3.2%	0.23	-0.74	-0.94
18	3.2%	0.24	-0.70	-0.89
19	3.3%	0.26	-0.65	-0.84
20	3.3%	0.27	-0.61	-0.83
21	3.3%	0.28	-0.57	-0.82
22	3.3%	0.30	-0.53	-0.79
23	3.3%	0.31	-0.49	-0.78
24	3.3%	0.33	-0.45	-0.78
25	3.4%	0.34	-0.41	-0.74
26	3.4%	0.35	-0.37	-0.73
27	3.5%	0.37	-0.34	-0.66
28	3.5%	0.38	-0.30	-0.63
29	3.5%	0.40	-0.26	-0.59
30	3.6%	0.41	-0.23	-0.52
31	3.7%	0.42	-0.19	-0.50
32	3.7%	0.44	-0.16	-0.42
33	3.7%	0.45	-0.12	-0.42
34	3.8%	0.47	-0.09	-0.35
35	3.9%	0.48	-0.05	-0.32
36	4.1%	0.49	-0.02	-0.12
37	4.1%	0.51	0.02	-0.11
38	4.2%	0.52	0.05	-0.01
39	4.4%	0.53	0.09	0.13
40	4.4%	0.55	0.12	0.14
41	4.4%	0.56	0.16	0.21
42	4.5%	0.58	0.19	0.25
43	4.5%	0.59	0.23	0.26
44	4.6%	0.60	0.26	0.33
45	4.6%	0.62	0.30	0.35
46	4.7%	0.63	0.34	0.48
47	4.8%	0.65	0.37	0.53
48	4.8%	0.66	0.41	0.54

Histogram	
Bin	**Frequency**
2%	0
3%	10
4%	25
5%	17
6%	18
7%	2
8%	0
More	0

49	4.9%	0.67	0.45	0.59
50	4.9%	0.69	0.49	0.64
51	4.9%	0.70	0.53	0.65
52	5.0%	0.72	0.57	0.67
53	5.0%	0.73	0.61	0.73
54	5.1%	0.74	0.65	0.81
55	5.3%	0.76	0.70	0.93
56	5.3%	0.77	0.74	0.95
57	5.3%	0.78	0.79	0.96
58	5.4%	0.80	0.84	1.07
59	5.5%	0.81	0.89	1.11
60	5.6%	0.83	0.94	1.26
61	5.6%	0.84	1.00	1.26
62	5.6%	0.85	1.05	1.27
63	5.8%	0.87	1.12	1.45
64	5.8%	0.88	1.18	1.45
65	5.9%	0.90	1.26	1.47
66	5.9%	0.91	1.34	1.47
67	5.9%	0.92	1.43	1.52
68	5.9%	0.94	1.53	1.54
69	5.9%	0.95	1.66	1.54
70	6.0%	0.97	1.82	1.59
71	6.2%	0.98	2.04	1.80
72	6.3%	0.99	2.46	1.89
Mean	4.2%			
St Dev	1.1%			
Count	72			
Skew	0.26			





Shapiro-Wilk test for normality

W = 0.92657, P-value=0.00047

p-value < 0.1, therefore reject null hypothesis - conclude data is not normally distributed

Computed a bootstrap confidence interval for the mean (computed in R statistical software)

Shapiro-Wilk test for normality of bootstrap means:

W=0.98938; p-value=0.145

p-value > 0.1, therefore null hypothesis cannot be rejected. Assume bootstrap means resampled data normally distributed.

Descriptive statistics (of bootstrap means):

Mean	4.2%	%
Median	4.2%	%
St Dev	0.3%	%
95% Conf. Interval (2.5%)	3.8%	%
95% Conf. Interval (97.5%)	4.8%	%

B.5 Case study operational and financial models

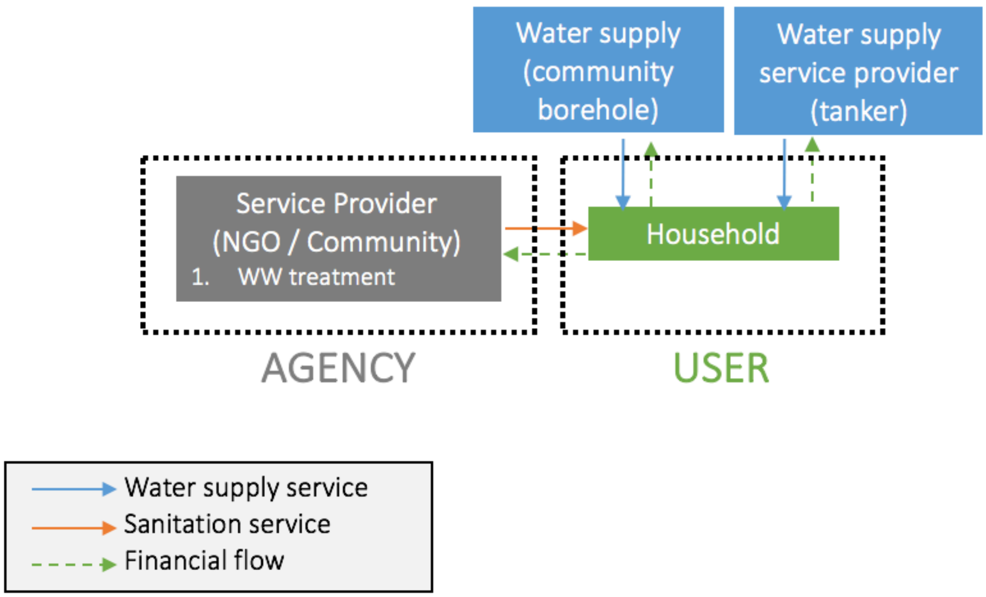


Figure B.17: Operational and financial models: Decentralized sewered, Zambia

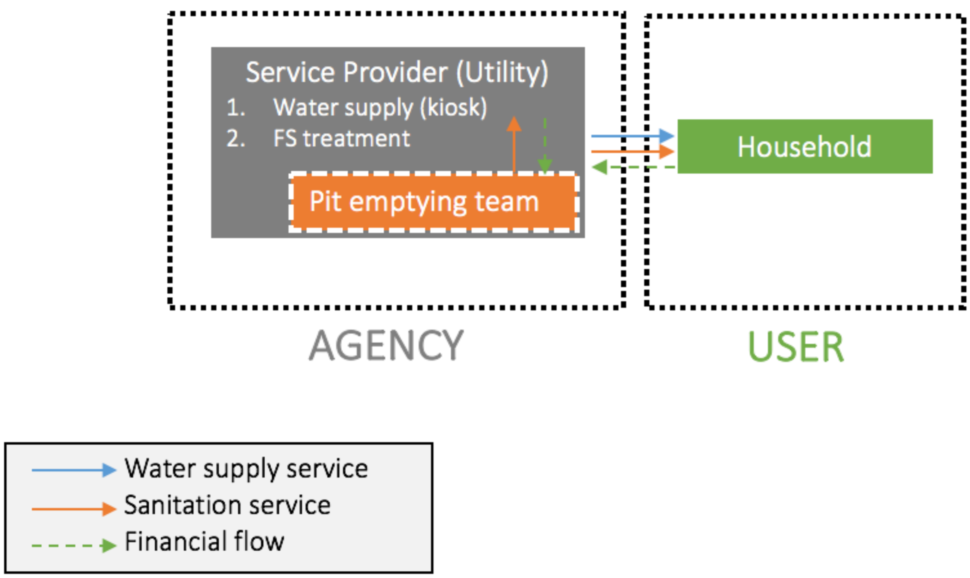


Figure B.18: Operational and financial models: Non-sewered, Zambia

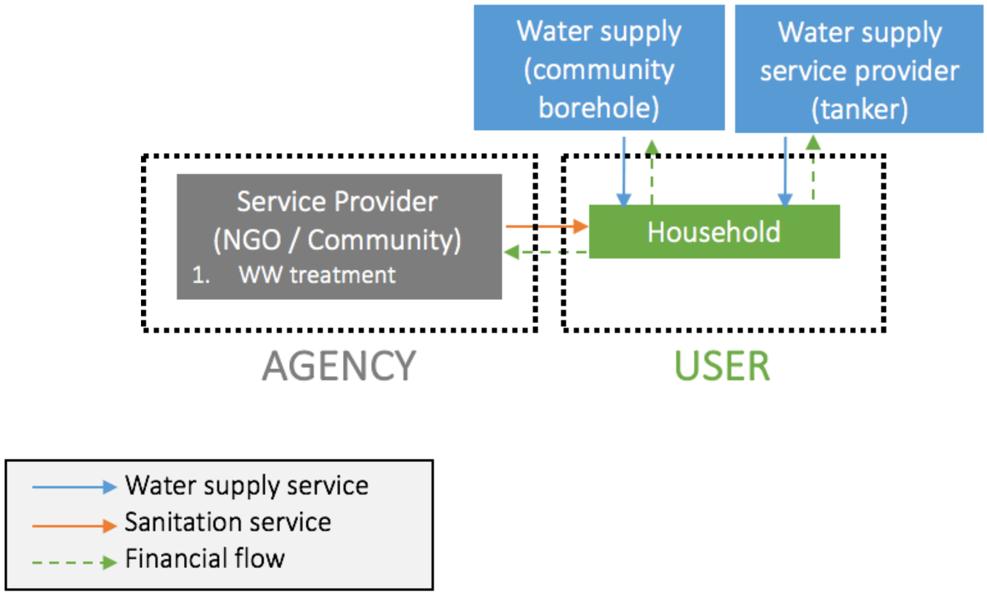


Figure B.19: Operational and financial models: Decentralized sewered, India

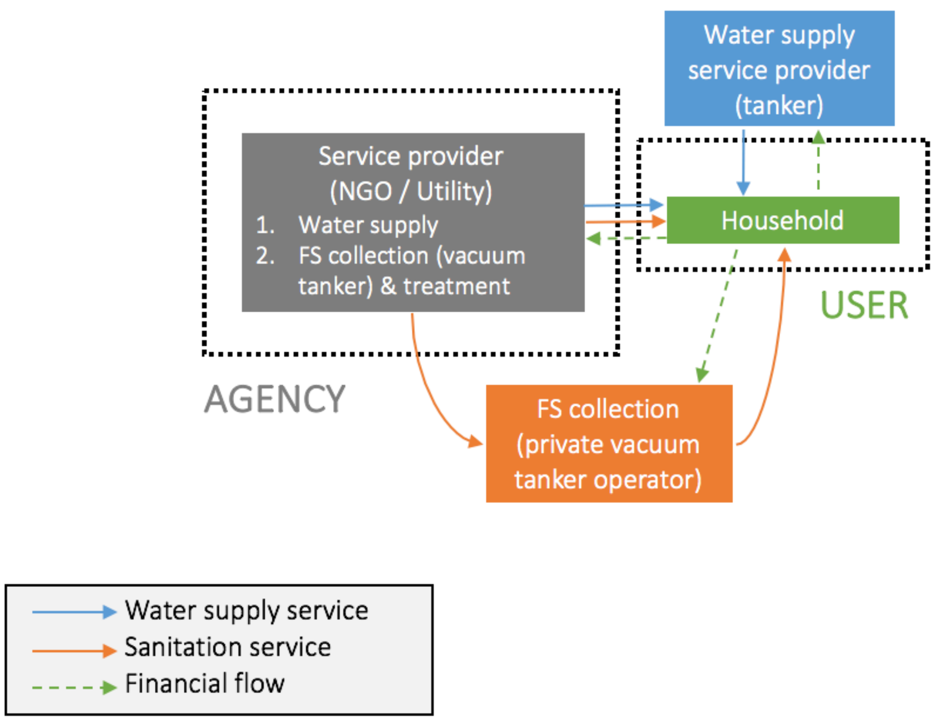


Figure B.20: Operational and financial models: Non-sewered, India

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