ENGINEERING IN THE TROPICS: EVALUATING A SOLAR-POWERED ANAEROBIC DIGESTION AND HYBRID CONSTRUCTED TREATMENT WETLAND SYSTEM TO TREAT AGRICULTURAL WASTES IN COSTA RICA

By

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ABSTRACT

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EVALUATING A SOLAR-POWERED ANAEROBIC DIGESTION AND HYBRID CONSTRUCTED TREATMENT WETLAND SYSTEM TO TREAT AGRICULTURAL WASTES IN COSTA RICA

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In Costa Rica, treatment of biomass residues and wastewater from agro-industrial and agricultural is often neglected and, consequently, Costa Rica has a severe water pollution problem. This study evaluated the performance of a solar-powered anaerobic digestion and hybrid constructed treatment wetland system to treat agricultural wastes in Costa Rica. The integrated solar thermal collector, anaerobic digester, and hybrid constructed treatment wetland system was implemented in 2013 at the Fabio Baudrit Experimental Station, Costa Rica and was proposed as a decentralized self-sufficient, close-loop, organic waste treatment system technology for rural areas.

A solar thermal collector unit provided sufficient energy to maintain thermophilic temperatures in a 20 m^3 anaerobic digester, which converted agricultural wastes, primarily food waste and chicken litter, to biogas and nutrient-rich solids. A constructed treatment wetland was used to treat water from the digestate for use in the digester and/or irrigation. The entire treatment system was self-sufficient, producing surplus energy. In general, 263 MJ of energy, 28 kg of fertilizer, and 550 kg of treated water were generated from 863 kg of mixed animal and food wastes. The net revenue considering electricity and fertilizer was $2,146 annually. The payback period for the system was estimated to be 21 years; however, a sensitivity analysis demonstrated that through optimization, the payback period could be reduced to 9 years.

The hybrid constructed treatment wetland achieved a treatment performance that allowed reuse of the treated water for other activities (e.g., irrigation and reuse at the digester). Pollutant removals were 99.5% for chemical oxygen demand (COD), 94.4% for total solids (TS), 99.8% for total nitrogen (TN), and 99.1% for total phosphorus (TP) during the rainy season and 96.4% for COD, 86.5% for TS, 98.9% for TN, and 99.6% for TP during the dry season. The hybrid configuration, a vertical subsurface flow and a free water surface constructed treatment wetland in series, was essential to overcome the individual weaknesses of each type of wetland, especially with regards to water storage and nitrogen. In addition, the vertical subsurface flow constructed treatment wetland did not become clogged after four years and a constant void space in the filter media of 20 of 30 m^3 was estimated from August 2015 to March 2016.

The system was exergetically sustainable with an environmental exergy efficiency ($\eta_{env,ex}$) of 5.60 and a total pollution rate ($R_{pol,ex}$) of -0.821, due to a positive exergy balance in which food waste and chicken litter were converted into high quality end products (i.e., energy, fertilizer, and treated water). Precipitation did not impact the exergy-based assessment of sustainability and more exergetically favorable $\eta_{env,ex}$ and $R_{pol,ex}$ values during the dry months were obtained due to better digestion performance, storage of water for future use, and biomass production. From the technical point of view, the system can contribute to sustainability of agricultural systems and communities in Costa Rica. This approach represents an academic effort toward waste treatment not only in Costa Rica, but also to other rural areas in the tropics.

For God,
for giving me the most precious gift in my life:
my parents.
Love you both

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Finally, to Noilin, who came at the end of this long process, love you.

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KEY TO ABBREVIATIONS

ACEPSA	Central American Association for Economy, Health and Environment
ADREC	Anaerobic Digestion Research and Education Center
AyA	Costa Rican Water and Sewage Institute
CapEx	Capital expenditures
CIA-UCR	Agronomy Research Center, University of Costa Rica
CELEQ	Center for Research in Electrochemistry and Chemical Energy
CHP	Combined heat and power
CICA-LCA	Water Quality Laboratory-Research Center of Environmental Pollution
CSTR	Continuous stirred tank reactor
CTW	Constructed treatment wetland
DAQ	Data acquisition system
ECA	Costa Rican Accreditation Institute
EEAFBM	Fabio Baudrit Experimental Station
FAO	Food and Agriculture Organization of the United Nations
FWS-CTW	Free water surface constructed treatment wetland
GDP	Gross domestic product
GHG	Greenhouse gas
HCTW	Hybrid constructed treatment wetland
HDPE	High-Density-Polyethylene
HSSF-CTW	Horizontal subsurface flow constructed treatment wetland
HLR	Hydraulic loading rate

HRT	Hydraulic retention time
ICE	Costa Rican Electricity Institute
IRET	Regional Institute for Studies on Toxic Substances
MINAET	Ministry of the Environment and Energy
MINSALUD	Ministry of Health
MACRS	Modified Accelerated Cost Recovery System
MSU	Michigan State University
OLR	Organic loading rate
OpEx	Operational expenditures
SPAD-HCTW	Solar-powered anaerobic digester-hybrid constructed treatment wetland
TEC	Technological Institute of Costa Rica
TIS	Tank in series
UASB	Up-flow anaerobic sludge blanket
UNA	National University
UCR	University of Costa Rica
VSSF-CTW	Vertical subsurface flow constructed treatment wetland

CHAPTER 1: INTRODUCTION

This chapter introduces the problem regarding biomass residues and wastewater in Costa Rica, the decentralized self-sufficient, close-loop, organic waste treatment system, and the scope and aim of this research. The problem regarding wastes in Costa Rica justifies technology selection: anaerobic digestion, solar thermal collectors, and constructed treatment wetlands. The integration of these technologies is presented and named as the solar-powered anaerobic digestion and hybrid constructed treatment wetland system or, simple, the SPAD-HCTW. The SPAD-HCTW is proposed as a decentralized self-sufficient, close-loop, organic waste treatment system for agro-industrial and agricultural activities in rural areas of Costa Rica. An overview of the SPAD-HCTW presents where the system was constructed, basis of design, and preview performance before this study. Finally, the configuration of the system studied in this dissertation and the scope and aims of this research are presented.

1.1. Biomass residues and energy production

Costa Rica is advancing toward carbon neutrality, with a goal of being carbon neutral by 2021. One action that reduces carbon emissions is to substitute fossil fuels with renewable energy sources (Coto 2013). In Costa Rica, it is estimated that agricultural and agro-industrial activities produce 6,000 metric tons of biomass residues per year. These residues could potentially be used to generated 600 MW of electricity (MINAET 2011, Coto 2013), or 23% of the installed electrical generation capacity of 2,600 MW in Costa Rica (Coto 2013). Currently, only 0.73% (18.9 MW) of Costa Rica's energy is produced from biomass residues, which indicates that a vast potential of energy is wasted (EN 2015).

Anaerobic digestion is a suitable technology for handling mixtures of biomass residues with high moisture content (65 – 99.9%), such as animal manures, food wastes, sewage wastewater, and other industry organic residues (IRENA 2012, Funk, Milford et al. 2013). Anaerobic digestion is a biological process performed by methanogenic communities that convert biomass residues into biogas in the absent of oxygen (Gould and Crook 2010) while generating solid and liquid digestate. Anaerobic digesters typically operate at different temperatures: psychrophilic (<25°C), mesophilic (25-40°C), and thermophilic (>45°C) (El-Mashad, Zeeman et al. 2004, Suryawanshi, Chaudhari et al. 2010). As reaction rates by microorganisms increase with increasing temperatures, digestion yields higher biogas production with higher temperatures. Thus, in contrast to mesophilic anaerobic digestion, thermophilic anaerobic digestion is characterized by higher microbial growth rates, higher solids reduction, improved solid-liquid separation, higher destruction of pathogens, and improved odor control (Zabranska, Dohanyos et al. 2002, Suryawanshi, Chaudhari et al. 2010). However, thermophilic anaerobic digestion requires higher input energy to maintain temperatures greater than 45°C.

To achieve thermophilic temperatures, combined heat and power (CHP) systems use a portion of the biogas produced to heat the digestate (Zabranska, Dohanyos et al. 2002). However, for small-scale digesters, biogas utilization for heating can result in negative energy balances (Vindis, Mursec et al. 2009), and for large-scale digesters, heating greatly reduces surplus energy (Zabranska, Dohanyos et al. 2002). Solar energy is an alternative, renewable energy source for heating digestion processes. In tropical regions, solar thermal collectors can satisfy the energy requirement necessary to maintain thermophilic temperatures year-round without any additional energy sources. Therefore, this study implemented flat-plate solar thermal collectors to provide heat for a thermophilic continuous stirred tank reactor (CSTR) anaerobic digester in Costa Rica.

1.2. Wastewater treatment and the sanitation problem

In Costa Rica only 5.81% of the 0.086 km^3/year of domestic wastewater is treated (Sato, Qadir et al. 2013). In addition, treatment of industrial wastewater is infrequent; for example, only 175 of 3,500 industries in the *Rio Virilla* watershed, which represents only 0.3% of Costa Rican territory, treat their wastewater (UN 2009, Coto 2013, Echeverria and Cantillo 2013). Therefore, it is clear that wastewater treatment of any source, including agricultural and agro-industrial activities, is frequently neglected. Consequently, untreated wastewater is likely impacting aquatic ecosystems, potentially causing health problems and environmental degradation (Dallas, Scheffe et al. 2004, UN 2015).

Wastewater treatment aims to reduce the level of pollutants in the wastewater before reuse or disposal into the aquatic ecosystems, thereby alleviating its negative impacts. Ecologically-based treatment systems tend to be less costly and sophisticated in operation and maintenance. For example, constructed treatment wetlands (CTWs) are biological systems with low capital cost and simple operation, with potentially efficient removal of pollutants; however, CTWs require preliminary treatment of the raw wastewater and demand relatively larger areas than centralized wastewater treatment facilities (the latter of which is usually not critical for rural areas) (Dallas, Scheffe et al. 2004). In this study, CTW technology is proposed to treat the liquid effluent from the thermophilic CSTR anaerobic digester in the tropics.

1.3. Self-sufficient energy production and waste treatment system

Even though biomass residues and wastewater pose an environmental concern to the aquatic ecosystems, both of them are rich in carbon and nutrients, representing a potential source of energy. Treating and utilizing the biomass residues and wastewater can lead to economic, social, and environmental benefits in rural areas (UN 2015). As stated by the Report of the Special Rapporteur on the human right to safe drinking water and sanitation, Catarina de Albuquerque, in 2013, “development must be sustainable and must protect the environment on which present and future generation depend” (UN 2013). One action is to constrain resource consumption and waste generation, to satisfy human needs. Therefore, an integrated biomass and wastewater treatment system that can provide energy, fertilizers, and reclaimed water could aid sustainable development for rural areas in the tropics. The solar-powered anaerobic digester and hybrid constructed treatment wetland system (SPAD-HCTW) combines solar thermal collection, anaerobic digestion, and CTW technologies to treat biomass residues and wastewater from agro-industrial and agricultural activities, while producing energy, fertilizers, and reclaimed water. Thus, the SPAD-HCTW can be offered as one approach for solving the problem of water pollution due to agro-industrial and agricultural waste disposal in rural areas of Costa Rica.

1.4. The solar-powered anaerobic digestion and hybrid constructed treatment wetland system

In 2011-2012, the SPAD-HCTW was designed and constructed. The design corresponded to knowledge and expertise developed in temperate regions with the intention of adapting the technology into the tropics. The system is located at the Fabio Baudrit Experimental Station (EEAFBM), in the province of Alajuela, Costa Rica (10.00 m N, -84.26 m W). The altitude of this site is 840 m above the level of the sea. The average annual precipitation is 1,940 mm, distributed from May to November, and the average annual temperature is 22°C (IMN 2016). EEAFBM is owned and operated by the University of Costa Rica (UCR) and promotes the holistic development of sustainable agro-industrial and agricultural activities in Costa Rica. One of EEAFBM's missions is to research innovative treatment/reutilization methods for biomass residues and wastewater. Therefore, this site was selected to house the SPAD-HCTW. Moreover, several organic wastes can be provided on-site (e.g., chicken litter and crop residues). In addition, EEAFBM is in a rural area, close to agro-industries and farms potentially interested in the technology (e.g., cow manure, coffee residues, and food wastes).

The SPAD consists of two integrated technologies that cover 180 m^2: the solar thermal collection unit and the anaerobic digestion unit. The HCTW consists of four constructed treatment wetlands, which covers an effective treatment area of 576 m^2 (Figure 1.1).

Figure 1. 1. The SPAC-HCTW located at EEAFBM, Alajuela, Costa Rica. Satellite image was taken from Google Earth.

The proposed system was designed based on previous studies at the Michigan State University (MSU) Anaerobic Digestion Research and Education Center (ADREC). The estimated mass and energy balance for the proposed system in Costa Rica was based on a study of a 20-L bench-scale solar-biopower system at MSU. The design of the HCTW followed Healy, Rodgers et al.(2007) and Kadlec and Wallace (2009) approaches for intermittent sand filters and wetlands for wastewater; however, the size of the HCTW was greatly reduced due to space constraints. The entire system was designed for the flow rate of one cubic meter per day. Table 1.1 shows the technical parameters of individual units of the system.

Table 1. 1. Description and basis of design.

Unit	Component	Estimated value	Current value at EEAFBM	Basis of design
Solar thermal collection	Flat-plate thermal collectors	50 m^2	36 m^2	50 m^2 based on an annual irradiance of 10 MJ/ m^2. Area was reduced to 36 m^2 based on new estimations on-site and area and budget constraints.
	Hot water tank	5 m^3	5 m^3	Stores heated water from the flat-plate solar thermal collectors.
	Hot water pump (UP 26-99 F from Grundfos, Olathe, KS)	n.d.	0.17 hp	Circulates water from the flat-plate thermal collectors to the hot water tank and vice-versa. Every day, a timer controls the hot water pump, which works from 9 am to 4 pm (solar radiation time at EEAFMB).
	Hot water pump (PB 351MA from Wilo, Korea)	n.d.	0.46 hp	Circulates hot water from the hot water tank through a High-Density-Polyethylene (HDPE) tubing heat exchanger to heat the digester and maintain the thermophilic temperature (45 ± 2°C).
	HDPE tubing heat exchanger	n.d.	40 m	Originally, a copper tubing heat exchanger was installed. This material was corroded by biogas inside the AD, thus HDPE tubing was chosen for replacing the broken copper tubing.
	Data acquisition system (CR1000 Campbell Scientific, Logan, UT)	--	--	Data acquisition system (DAQ) collects data from thermocouples (type K, probe ungrounded) within the digester every 20 seconds for feedback control of the digestion temperature (±2°C of the set temperature of 45°C). The DAQ sends a digital signal to power the hot-water pump if the digester temperature was lower than the set temperature.

Table 1. 1. (cont'd)

Unit	Component	Estimated value	Current value at EEAFBM	Basis of design
Thermophilic CSTR anaerobic digester	Anaerobic digester tank	Volume: 20 m^3 Flow rate: $1m^3/d$ HRT: 10-15 days Mixture characteristics TS: 10% COD: 90 g/L	Volume: 20 m^3 Flow rate: $1m^3/d$ HRT: 20 days	Cylindrical tank with flat bottom made with HDPE. A submersible digester mixing pump (model 5763 from ATM, Royersford, PA) operates 10 minutes each hour to assure homogeneous conditions in the digestate. TS was set to 2% based on desired biogas production of 20 m^3/d.
	Feeding tank	n.d.	10 m^3	Cylindrical tank with flat bottom made with HDPE. Stores the mixture to be fed into the digester. Weekly, five cubic meters of mixture is prepared with chicken litter and food waste at an average ratio of 1:12 (dry mass basis) and mixed with five cubic meter of reclaimed water from the HCTW. This ratio was selected to assure 2.2% of total solids in the mixture. The feeding tank has an external pump (model AMTP/N 1626-305-00 from ATM, Royersford, PA) for mixing (six minutes before feeding) and feeding the mixture into the anaerobic digester.
	Effluent storage tank	n.d.	10 m^3	Cylindrical tank with flat bottom made with HDPE. Stores liquid digestate. Has sufficient storage for two-weeks' worth of digestate.
	Biogas bag	60 m^3	60 m^3	Biogas bag made with HDPE. Stores biogas for engine use to maintain a steady power and heat flow. Size was selected based on biogas production. Initial calculation estimated a biogas production of 20 m^3/d.
Electric generator	Combined heat and power (CHP) engine	One engine 30 kW	Two engines 16 kW each one	Power was chosen based on the initial energy estimation of 684 MJ/d. CHP engine (Branco® B4T-5000 Bioflex, Brazil) powers pumps and other pieces of equipment in the system to satisfy operational requirements.

Table 1. 1. (cont'd)

Unit	Component	Estimated value	Current value at EEAFBM	Basis of design
Constructed treatment wetland (CTW)	Area	1,100 m^2 Inlet flow rate: 1m^3/d. COD inlet concentration: 45,000 mg/L.	576 m^2	Area of 1,100 m^2 was calculated using the tank in series (TIS) model (Kadlec and Wallace 2009) for a free water surface constructed treatment wetland. Maximum effluent concentrations are the discharge standards for Costa Rica (COD < 150 mg/L, TS < 50 mg/L, TN < 50 mg/L, TP < 8 mg/L, 5 < pH < 9, and 15 °C < T < 40 °C) (MINAE-MSP 2007). Final area of 576 m^2 was chosen due to space constraints and research goals at EEAFBM.
	Hybrid constructed treatment wetland	n.d.	4 cells.	Cell 1: Intermittent sand filter. Cell 2: Vertical subsurface flow constructed treatment wetland (VSSF-CTW). Cells 3 and 4: free water surface constructed treatment wetland (FWS-CTW) planted with floating plants. Dimensions. Top: 12 x 12 m square. Bottom: 9 x 9 m square. Depth: 1.1 m. Walls' slope: 27° with respect to the horizontal. All cells are interconnected, and each cell has a submersible pump (WS V52 from Franklin Electric, Fort Wayne, IN).
	Cells 1 and 2	--	--	Media profile, from bottom to top: 0.2 m of stone (particle size of 12-20 mm), 0.2 m of pea gravel (particle size of 4-8 mm), and 0.7 m of coarse sand (particle size of 0.75-2 mm, 32% porosity). Media profile and particle size were chosen based on Healy, Rodgers et al. (2007).
	Cell 2 planting selection	*Juncus effusus* *Coix Lacryma jobi* *Cyperus papyrus* *Iris graminea* *Canna indica*	-- -- *Cyperus papyrus* *Iris graminea* *Canna indica*	Plants were planted in March 2013. Five emergent plants were chosen based on availability on-site and literature review of wetlands in the tropics. Only *Cyperus papyrus*, *Iris graminea*, and *Canna indica* were considered in this study as the other two plants did not s survive in the wetland.
	Cell 2 recirculation	2:00 am to 4:00 pm Height: 1.60 m 4 upright sprinklers	2:00 am to 4:00 pm Height: 1.60 m 4 upright sprinklers	Recirculation at night to avoid higher losses of water due to higher evaporation rates during the day. Height of the recirculation spray was chosen to decrease interference of the redistribution spray by plants. Sprinklers were located at each corner of a square (7.00 × 7.00 m) centered at the center of the wetland to reach maximum wetland surface area.

Table 1. 1. (cont'd)

Unit	**Component**	**Estimated value**	**Current value at EEAFBM**	**Basis of design**
(CTW)	Cell 2 geotextile membrane	n.d.	2 x 2 x 0.25 m^3 (length x width x depth)	Dimensions were chosen based on the discharge of 1 m^3 into the wetland.
	Cell 4 planting selection	*Eichornia crassipes* *Pistia stratiotes* *Salvinia*	*Eichornia crassipes* *Pistia stratiotes* --	Plants were planted in March 2013. Three floating plants were chosen based on availability on-site and literature review of wetlands in the tropics. Only *Eichornia crassipes* and *Pistia stratiotes* were considered in this study as *Salvinia* was no longer present in August 2015.

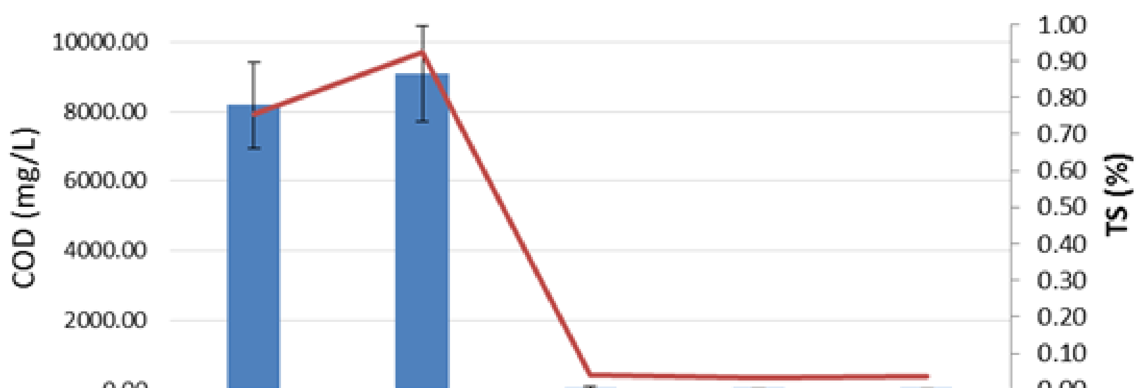
** TS: total solids

* COD: chemical oxygen demand

1.5. Overview of the system evaluated in this study

The SPAD-HCTW began operation in March 2013. One cubic meter of food waste and animal manure was fed into the anaerobic digester and one cubic meter of digestate was discharged into the HCTW daily. Initially, the HCTW operated in series, from cell 1 to cell 4, transferring one cubic meter per day between cells. The liquid effluent was discharged into cell 1 or cell 2 due to high concentration of chemical oxygen demand (COD), total solids (TS), and ammonium (NH_4). After observation of leaking in cell 1 during the first few months of operation, digestate was only discharged into cell 2. The goal of the first two cells was to remove large portions of COD and TS while reducing NH_4 concentrations. Intermittent sand filters and vertical subsurface flow constructed treatment wetlands (VSSF-CTWs) provide oxidative conditions for degradation of solids trapped in the filter media and transformation of ammonium to nitrate. Treated water from cell 2 was transferred to cell 3, then to cell 4, for polishing. Free water surface constructed treatment wetland (FWS-CTW) provides anaerobic zones for denitrification to reduce concentrations of nitrate (NO_3).

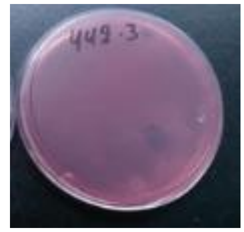
The SPAD-HCTW has operated for 30 months, from March 2013 to July 2015. During this period, little data on treatment performance was collected. In 2014, COD and TS were measured at the HCTW (Figure 1.2) and fecal coliforms were evaluated (Table 1.2). In December 2014, Lambda Laboratories (certified by the Costa Rican Accreditation Institute (ECA)) conducted a sampling campaign on the entire system (Table 1.3). Results indicated that a reduction of solids (56% of suspended solids and 36% of total organic carbon) was occurring in the anaerobic digester (Table 1.3). These data were inconsistent with the data shown in Figure 1.2, where COD and TS were not removed by the digester. Consistently, Figure 1.2 and Table 1.3 show that cell 2, the VSSF-CTW, removed large portions of TS, COD, total organic carbon, and nutrients (as total nitrogen (TN) and total phosphorus (TP)); while, the cell 3 and cell 4, the FWS systems, polished the treated water. The evaluation of fecal coliforms indicated that the treated water was pathogen free, except for a single detectable result at the effluent from the anaerobic digester in October 2014.



	Reactor (S1)	Effluent (S3)	Inlet cell 3 (S4)	Inlet cell 4 (S5)	Outlet cell 4 (S6)
	8183.33	9091.25	53.33	27.75	16.83
TS	0.76	0.92	0.04	0.03	0.04

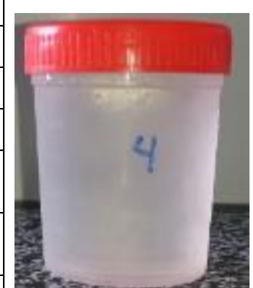
Figure 1. 2. Removal of COD and TS through the system. Reactor (S1): sampling of the mixture before feeding. Effluent (S3): sampling of the effluent after digestion. Inlet cell 3 (S4): sampling of the effluent from cell 2 entering cell 3. Inlet cell 4 (S5): sampling of the effluent from cell 4 entering cell 4. Outlet cell 4 (S6): sampling of the effluent from cell 4.

Table 1. 2. Evaluation of fecal coliforms after digestion.

Sampling point	Dates 22/10/14 MPN	3/12/14 CFU/mL	10/12/14 CFU/mL	
Digester effluent (S3)	Detectable	Non-detectable	Non-detectable	
Effluent cell 2/ Inlet cell 3 (S4)	Non-detectable	5.1 x 10^3	Non-detectable	
Effluent cell 3/ Inlet cell 4 (S5)	Non-detectable	3.2 x 10^2	Non-detectable	
Outlet cell 4 (S6)	Non-detectable	Non-detectable	Non-detectable	

Notes. MPN: Most Probable Number. CFU: Colony-forming unit. Effluent (S3): sampling of the effluent after digestion. Inlet cell 3 (S4): sampling of the effluent from cell 2 entering cell 3. Inlet cell 4 (S5): sampling of the effluent from cell 4 entering cell 4. Outlet cell 4 (S6): sampling of the effluent from cell 4.

Table 1. 3. Water quality analysis of entire system from Lambda Laboratory (Dec. 2014).

Sampling points	TOC (mg/L)	SS	TN	TP	
Allowed by law in Costa Rica	150	50	50	25	
Reactor (S1)	6,460 ± 130	13,840 ± 6	1,750 ± 40	107 ± 1	
Digester effluent (S3)	4,140 ± 80	6,200 ± 6	1,675 ± 40	73 ± 1	
Effluent cell 2/ Inlet cell 3 (S4)	13 ± 1	62 ± 6	5.06 ± 0.5	0.10 ± 0.02	
Effluent cell 3/ Inlet cell 4 (S5)	8.5 ± 0.5	26 ± 6	2.8 ± 0.5	0.04 ± 0.02	
Outlet cell 4 (S6)	9.0 ± 0.5	12 ± 6	3.5 ± 0.5	> 0.02	

Note. Reactor (S1): sampling of the mixture before feeding. Effluent (S3): sampling of the effluent after digestion. Inlet cell 3 (S4): sampling of the effluent from cell 2 entering cell 3. Inlet cell 4 (S5): sampling of the effluent from cell 4 entering cell 4. Outlet cell 4 (S6): sampling of the effluent from cell 4. TOC: Total organic carbon. SS: Suspended solids. TN: Total nitrogen. TP: Total phosphorus.

1.6. The system studied in this dissertation

This study evaluated the SPAD-HCTW in detail from August 2015 to March 2016. This study represented the first campaign of continuous (weekly) monitoring of the system. Herein, the HCTW consisted of the VSSF-CTW, cell 2, and the FWS-CTW, cell 4. Budget constraints limited the inclusion of the entire system (4 cells). However, the VSSF-CTW working in-series with the FWS-CTW allowed the evaluation of treatment performance and clogging. A description of the SPAD-HCTW is shown in Figure 1.3. The CSTR anaerobic digester was assisted by flat-plate solar thermal collectors to maintain thermophilic conditions during digestion. Produced biogas was used as a fuel for running two electric generators (16 kW each), which produced electricity for on-site uses. Solid digestate was composted and used for fertilizing the crops at EEFBM. Liquid digestate was further treated by the HCTW, which included two CTWs working in-series, a VSSF-CTW followed by a FWS-CTW. Treated water was used for irrigation, reused in the anaerobic digester, or discharged into surface water.

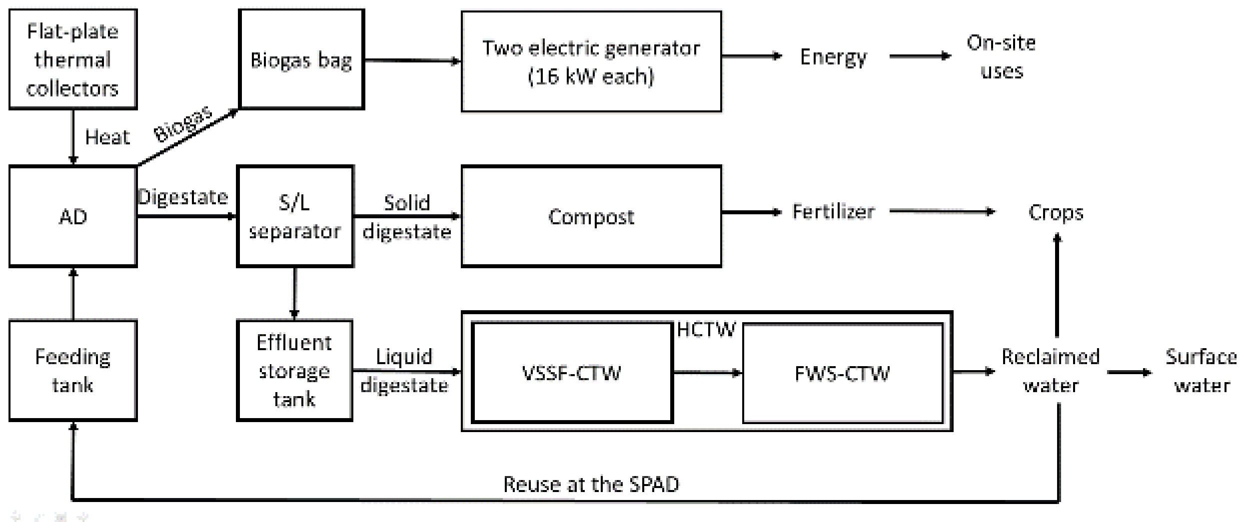


Figure 1. 3. Conceptualization of the SPAD-HCTW system. Abbreviations: AD: Anaerobic digester; S/L separator: Solid/liquid separator; VSSF-CTW: Vertical subsurface flow constructed treatment wetland; FWS-CTW: Free water surface constructed treatment wetland; HCTW: Hybrid constructed treatment wetland.

1.7. Scope and aims of this research

Treatment of biomass residues and wastewater from agro-industrial and agricultural activities by the SPAD-HCTW is proposed as a decentralized self-sufficient, close-loop, organic waste treatment system technology for rural areas at the tropics. Therefore, the scope of this dissertation is to evaluate the treatment performance and exergetic sustainability of the SPAD-HCTW. In particular, the following principal research question will be addressed: *Does the SPAD-HCTW sustainably convert biomass residues and wastewater into energy, fertilizers, and treated water, while meeting standards for waste discharges into the environment?* The following four main questions were developed to answer the principal research question.

First, *is the SPAD a self-sufficient energy production and treatment system for providing energy and fertilizers from biomass residues, and treated water?* To answer this question, the integrated SPAD with a VSSF-CTW was evaluated in terms of technical performance. In particular, this study evaluated the technical performance of: 1) solar thermal collectors for maintaining thermophilic temperature in the anaerobic digester; 2) thermophilic anaerobic digestion for converting organic wastes into energy and fertilizers; and 3) VSSF-CTW for treating wastewater (i.e., liquid digestate). These evaluations were important to demonstrate that the SPAD is a self-sufficient, close-loop, energy production and waste treatment system. In addition, a cash flow analysis was conducted to determine the payback period of the current system. Technical performance of the solar-powered waste utilization and treatment system in Costa Rica is presented in Chapter Three.

Secondly, *how does the performance of the HCTW respond to precipitation conditions at EEAFBM?* Better treatment performance is expected during the precipitations due to dilution of pollutants in the wetlands. To answer this question, the treatment performance of the HCTW during rainy and dry periods at EEAFBM was analyzed. In addition, the applicability of loading charts and contaminant removal models commonly used to describe temperate wetlands to model treatment performance of the tropical HCTW was evaluated. Performance of the hybrid constructed treatment wetland is presented in Chapter Four.

Thirdly, *are implemented preventive strategies in the VSSF-CTW positively impacting in the longevity of the filter media by decreasing clogging?* Clogging of VSSF-CTWs is the largest challenging to maintaining a high performing system that can effectively treat wastes with high solids concentrations. Strategies to prevent clogging that were implemented in the VSSF-CTW were wastewater pretreatment and inlet distribution of the wastewater. Wastewater pretreatment included a rotary liquid/solid separation unit, an effluent storage tank that allowed settling, and a geotextile membrane. Influent was discharged into a geotextile membrane, which both distributed the influent and further filtered out solids. Finally, recirculation of treated wastewater distributed the partially treated water over the entire treatment area. To answer this question, the longevity of the sand media in the VSSF-CTW was evaluated for solid accumulation, root development, and infiltration. In addition, laboratory-scale VSSF-CTW was built to determine how different root structures affect clogging. These evaluations were important to determine practices that expand the lifespan of the VFSS-CTW. The clogging in tropical vertical subsurface flow constructed treatment wetlands is presented in Chapter Five.

Finally, *is the SPAD-HCTW exergetically sustainable technology?* This research question was answered by conducting an exergy-based assessment of sustainability using the environmental exergy efficiency ($\eta_{env,ex}$) and the total pollution rate ($R_{pol,ex}$) indexes. A system that converts materials with high entropy into high quality end products with low entropy will be sustainable if the balance between inputs (e.g.: wastes) and outputs (e.g.: energy, fertilizers, and treated water) is positive (Wall 2010, Woudstra 2016). Assessments were conducted for the exergy baseline or the SPAD alone (case 0), the SPAD and the VSSF-CTW (case 1), and the SPAD-HCTW (case 2) to compare potential improvement of sustainability through inclusion of wetlands during treatment of agricultural wastes. The exergy-based assessment of sustainability of a solar-powered anaerobic digestion and hybrid constructed treatment wetland system to treat agricultural wastes in Costa Rica is presented in Chapter Six.

Conclusions, limitations, and future work, in relation with the SPAD-HCTW as a decentralized wastewater treatment facility for rural areas in the tropics, are presented in Chapter Seven.

CHAPTER 2: LITERATURE REVIEW

This chapter introduces biomass residues and wastewater production from agro-industrial and agricultural activities in Costa Rica. This review reveals that the treatment of biomass residues and wastewater is frequently neglected, even though the existing legal framework in Costa Rica recognized the need for wastewater treatment. Consequently, improper management of biomass residues and wastewater causes a severe sanitation and water pollution problem in Costa Rica. To overcome the pollution problem, a discussion of technologies applicable to treat biomass residues and wastewater from agro-industrial and agricultural activities is presented here. Finally, exergy is introduced as a metric to evaluate the sustainability of new systems.

2.1. The opportunity: from linear to circular thought

The Food and Agriculture Organization of the United Nations (FAO) has defined a new approach for supporting the food security and sustainable agriculture: The Water-Energy-Food Nexus (FAO 2014). These three resources are essential for the human well-being and a positive resource balance between the supply from the environment and demand by the human is essential for sustainability. However, the balance tends to be negative due to the population growth and climate change, which in turn restrict even more the access to water, energy, and food. First, there is an unsustainable demand of water, energy, and food by the population based on a linear economic model (use, make, dispose). Consequently, the disposal of untreated residues degrades the environment and creates a severe pollution problem. For example, it is estimated that over 80% of wastewater worldwide goes untreated, so that more than two million cubic meters of wastewater are discharged into the world's rivers, lakes, and oceans every day

(Otchet 2003, UNESCO 2015). On the other hand, the effects of climate change, such as droughts, floods, and increasing temperatures, threaten the availability of resources (water, energy, and food) and restrict access to them for vulnerable populations. FAO (2014) estimates a world population of 10 billion people for 2050, which will result in a 60% increase in demand for food. Thus, the agro-industry and agricultural activities will require more water and energy, which currently account for, on average, consumption of 70% of fresh water and 30% of energy resources worldwide. Under this paradigm, there is an urgent need to reduce pressure on water, energy, and food resources. Sustainable development requires recognizing the linkages between these three resources (Biggs, Bruce et al. 2015). Once the linkages are recognized, integrated treatment systems can be implemented to optimize resource supply and consumption by the creation of close-loop systems to convert linear economies into circular economies (Agrocycle 2017). In contrast to a linear economy, a circular economy considers residues as resources, alleviating the demand of resources and reducing waste generation and disposal (Figure 2.1).

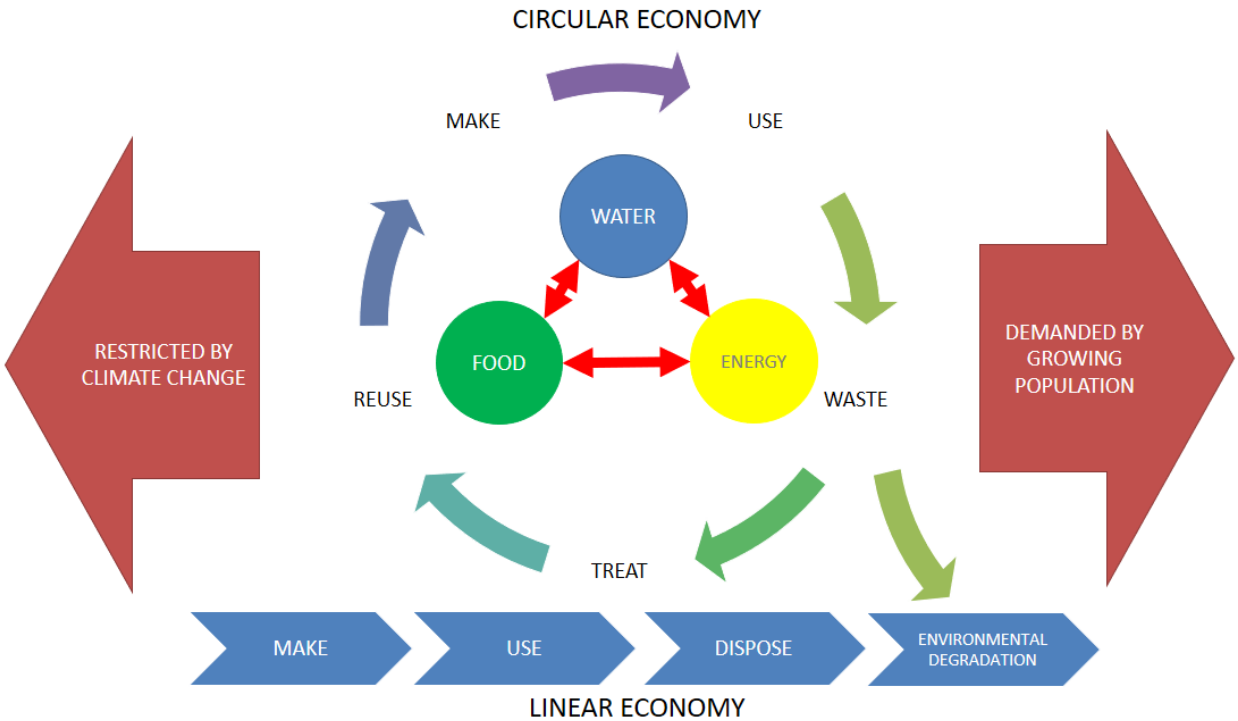


Figure 2. 1. Dynamics of the water-energy-food nexus in the economy (Own creation).

2.2. Biomass residues and wastewater from the agro-industry and agricultural activities in Costa Rica

In general, agro-industry and agricultural activities follow a linear economy model, with the simple disposal of biomass residues and wastewater adversely impacting the environment. In this section, the biomass residues and wastewater from agro-industry and agricultural activities in Costa Rica are characterized. Definition, characteristics, and estimated waste production are indicated for biomass residues and wastewater. Moisture content and gross calorific value are indicated for the solid wastes; whereas, water quality parameters are indicated for liquid wastes. In addition, regulations, efforts, and limitations are established for identifying the problem in Costa Rica.

2.2.1. Biomass residues from agro-industry and agricultural activities

In Costa Rica, biomass residues from agro-industry and agricultural activities are named RAOs (from the Spanish Residuos Agricolas Organicos) (Coto 2013). Biomass residues from agricultural crops (53% of total agricultural residues) are predominantly from sugar cane, pineapple, oil palm, coffee, and banana; while, biomass residues from livestock activities (47% of agricultural residues) are largely consist of cow manure, chicken litter, and swine manure. In addition, sawdust, wood chips, and other rejected wood products are sawmill residues, but only account for a small percentage of residues. Table 2.1 characterizes biomass residues based on the gross calorific value and moisture content. The gross calorific value indicates the potential energy (as heat) that can be obtained per kilogram of biomass residues when combusted. For the selected biomass residues, the gross calorific values are similar; however, moisture contents of the residues are substantially different. Thus, selection of technologies for treating and extracting energy from biomass residues should be based on moisture content. Technologies to convert biomass residues into energy include direct combustion, gasification, and anaerobic digestion (Zafar 2016). Direct combustion and gasification are appropriate technologies for low-moisture content feedstocks (IRENA 2012, Funk, Milford et al. 2013). For example, commonly used biomass residues for direct combustion are sawdust, wood bark, shavings, end cuts, and chips, hog fuel, bagasse, and rice husks – all materials with moisture contents of 10 – 50% on a wet-basis. Predominant biomass residues for gasification are wood chips, hog fuel, rice hulls, dried sewage sludge, pellets, wood scrapes, and nut shells, which have moisture contents ranging 15 – 50% (IRENA 2012, Funk, Milford et al. 2013). Alternatively, anaerobic digestion is an appropriate technology for processing biomass residues with moisture contents of 65 to 99% (IRENA 2012, Funk, Milford et al. 2013). In this study, animal manure and food waste are the biomass residues of interest. Coto (2013) estimates a production of 2,652,143 dry metric tons per year of animal manure. Thus, in Costa Rica, a total net energy of 349 MW electricity per year could be generated from animal manure, demonstrating the great potential Costa Rica has for electric energy generation from biomass residues. In comparison, the status quo capacity for electrical generation in Costa Rica is 2,600 MW.

Table 2. 1. Characteristics of selected biomass residues in Costa Rica.

Agricultural activities	**MC* (%)**	**Gross calorific value (MJ/kg)**	**Livestock activities (manure)**	**MC* (%)**	**Gross calorific value (MJ/kg)**	**Sawmill**	**MC* (%)**	**Gross calorific value (MJ/kg)**
Coffee pulp	81.0	15.9	Pig	85.0	13.8	Sawdust	32.0	18.5
Mucilage	81.0	15.9	Chicken	36.0	15.9	Woodchips	50.0	18.5
Coffee husk	11.0	17.9	Milk	80.0	15.6	Rejected wood (e.g., bark)	55.0	18.5
Rice husk	15.0	15.4	Meat	80.0	15.6	--	--	--
Sugar cane bagasse	50%	17.5	--	--	--	--	--	--
Field residues from pineapple	90%	11.6	--	--	--	--	--	--
Pineapple crown	78.5	11.6	--	--	--	--	--	--

*. MC: moisture content. Modified from Coto (2013).

With regards to regulatory agencies in Costa Rica, the Ministry of Health (MINSALUD) regulates public health while the Ministry of the Environment and Energy (MINAET) regulates environmental protection. These two institutions have developed the legal framework for management of biomass residues. In total, five laws and regulations have been decreed concerning legal aspects to the management of biomass residues. In general, three laws (General Health law, Organic Environmental law, and Law of Integrated Waste Management) ensure public health by the appropriate management of wastes; however, there is not a specific law or regulation for biomass residues from agro-industrial and agricultural activities. Instead, household residues (garbage and rubbish such as bottles, cans, clothing, compost, disposables, food packaging, food scraps, newspapers and magazines, and yard trimmings) and hazardous wastes are specific regulated (Table 2.2).

Table 2. 2. Laws and regulations for the management of solid residues in Costa Rica.

Law	**Main goal**	**Legal aspects with respect to biomass residues**
General Health law (N° 5395, 1973)	To ensure the public health of the population.	Forbid pollution of ground and surface water due to direct or indirect discharge of solid, liquid and gaseous wastes. Forbid discharge of industrial residues into the sewage system or storm sewer system. Industries should provide treatment of residues to avoid environmental pollution.
Organic Environmental law (N° 7554, 1995)	To ensure proper management of natural resources. To ensure environmental protection.	Promote plans for protection of the environment. Includes appropriate management of solid residues. Industries are responsible to provide treatment of residues to avoid environmental pollution.
Landfill regulation (N° 27387-S, 1995)	To regulate management and disposal of household residues.	Note: Does not include biomass residues and wastewater from agro-industrial and agricultural industries.
Law of Integrated Waste Management (N° 8839, 2010)	To regulate efficient use of resources and integrated management of wastes through the planning and execution of regulatory, operational, financial, administrative actions, and educational, environmental, and health monitoring and evaluation.	Industries should establish programs for efficient use of resources and management of solid wastes prior to disposal. Efficient use of resources includes reduce, recycle, and reuse strategies for reducing quantity of generated waste. Management of wastes includes treatment of residues to convert them into resources with added value.
General Regulation for the Classification and Management of Hazardous Waste (N° 37788-S, 2013)	Regulate management and disposal of industrial hazard residues.	Note: Does not include biomass residues and wastewater from agro-industrial and agricultural industries.

Instead of separate laws, biomass residues from agro-industrial and agricultural activities are regulated through this legal framework. Major efforts have been focused in the production of energy through the treatment of biomass residues. The *Non-conventional renewable energy* program, led by the Costa Rican Electricity Institute (ICE), has been promoting anaerobic digestion for the treatment of animal manure with the main goal of producing energy. Currently, renewable energy production from biomass represents 0.73% of the total power capacity in Costa Rica (18.9 of 2,600 MW) (ICE 2016). Based on estimates by Coto (2013), current energy production from biomass is only 5.44% of the potential energy that can be produced from animal manure in Costa Rica. Additionally, ICE also promotes anaerobic digestion projects as part of integrated watershed management for protecting reservoirs from hydroelectric energy production, which is the main renewable energy source in Costa Rica (75%, 1,950 of 2,600 MW) (ICE-UEN 2010). Prestigious academic institutions in Costa Rica (e.g., EARTH University, National University (UNA), University of Costa Rica (UCR), and Technological Institute of Costa Rica (TEC)) have played an important role in the treatment of biomass residues. These institutions have been developing research and extension projects promoting anaerobic digestion (Kinyua 2015). Unfortunately, there are no data bases of anaerobic digestion projects demonstrating the technical and treatment performance. A survey of biogas production elaborated by the ICE in 2014 simple indicates that 58% of the digesters are tubular, plug-flow bags (e.g., tubular polyethylene bag digesters) for small-scale farms; whereas covered lagoon anaerobic digesters correspond to 42%. The limited information that is available is from projects developed by ICE at many agro-industries and agricultural farms. Cow manure is the most utilized feedstock for biogas production, with a variable production (20 – 40 m^3/d), whereas, swine manure is an attractive alternative feedstock due to high energy production. In general, all these projects target to mesophilic conditions (Table 2.3).

Table 2. 3. Summary of anaerobic digester projects developed by the Non-conventional renewable energy program (ICE).

Project	Type of reactor	Volume (m^3)	Heads	Feedstock: Manure source	Biogas produced (m^3/d)	CH4 (%)	Power capacity (kW)
Robago Farm [1]	Covered lagoon	90.0	85	Cow	20	n.a.	30
Julieta Farm [1]	Covered lagoon	106.2	50	Cow	30	n.a.	38
Cerro Grande Farm [1]	Covered lagoon	44.0	50	Cow	21	66.0	20
Noble Farm [1]	Covered lagoon	86.4	120	Cow	40	n.a.	n.a.
Sermide Farm [1]	Covered lagoon	988	4,000	Swine	320	n.a.	60
El Cerro Farm [1]	Covered lagoon	2,093	5,530	Swine	412	80.0	70
Agro-Turistica Don Carlos Farm [1]	Covered lagoon	180	n.a.	Cow (with cheese whey)	29	62.0	n.a.
Porcina America [1]	Covered lagoon	3,600	30,000	Swine	1,800	60.0	250
Kafur Farm [1]	Covered lagoon	n.a.	4,000	Swine	n.a.	n.a.	70
EARTH University (Lansing, Víquez et al. 2008)	Tubular polyethylene bag digesters	85	5	Cow	27.5	62.6	40
EARTH University (Lansing, Víquez et al. 2008)	Tubular polyethylene bag digesters	85	40	Swine	6.0	76.4	40
Monteverde, Puntarenas (Kinyua 2015)	Tubular polyethylene bag digesters	12	10	Swine	2.83	71.0	n.a.

[1] Projects developed by the *Non-conventional renewable energy* program (ICE).
n.a. represents not assessed

Implementation of anaerobic digestion provides economic benefits to farms while satisfying legal compliance with Costa Rican regulations. These farms are saving money by the consumption of self-produced electric energy for operation of the farms or, for smaller projects, for cooking (Coto 2013, Kinyua 2015, ICE 2016). In addition, implementation of anaerobic digestion increases the electric energy generation from biomass in Costa Rica. Costa Rica is striving to be the first carbon-neutral country by 2021 and utilization of biomass residues as a renewable energy has been prioritized to reduce electricity production from fossil fuels.

Through conversion of organic matter into biogas, anaerobic digestion reduces organic matter and suspended solids within the digester. However, as the influent feedstock has very high concentrations of organic matter and suspended solids (e.g., >1 g/L concentrations), effluent digestate still has high solid, carbon, and nutrient contents (Lansing, Víquez et al. 2008, Kinyua 2015). For example, Lansing, Víquez et al. (2008) reported 796 and 189 mg COD/L, 1,440 and 717 mg TS/L, 178 and 177 mg TKN/L, and 16.6 and 19.8 mg TP/L, in the digester effluent from a dairy and swine farm, respectively. These values are much greater than the discharge standards in Costa Rica, discussed in the next section. Usually, digestate is applied to agricultural fields as a fertilizer for enhancing nutrient retention in soils. However, improper management of nutrients through land application can cause eutrophication (Sutton and Joern 1992, Johnson and Eckert 1995). In addition, mesophilic anaerobic digestion does not eliminate pathogens to concentrations below the detection limits; thus, there is potential health impact due to the transmission of pathogens to food and water when digestate from mesophilic reactors is land applied. For example, Kinyua (2015) detected *Giardia lamblia* and *Cryptosporidium parvum* in the effluent from a swine tubular polyethylene bag digester in Monteverde, Puntarenas.

2.2.2. Wastewater from agro-industry and agricultural activities

The Costa Rican Regulation of Dumping and Reuse of Wastewater law (N° 33601-S, 2007) defines wastewater as water that has been used and, due to the use, it characterized by the presence of pollutants. In addition, the regulation classifies wastewater as ordinary or special. Ordinary wastewater is wastewater from households, including grey and black wastewater;

whereas, special wastewater results from non-household sources. Thus, wastewater from agro-industry and agricultural activities is classified as special wastewater. In 2013, the MINSALUD had registered 5,028 industries producing special wastewater. However, only 30% of the industries presented an operational report, which is requested by the Regulation of Dumping and Reuse of Wastewater law (N° 33601-S, 2007). And, even worse, a small percentage of these reports are reliable. Thus, there is not an accurate estimation of the special wastewater produced in Costa Rica.

The characteristics of special wastewater widely differ based on source. For example, wastewater from crop production, food processing, slaughterhouses, livestock production, and anaerobic digesters have entirely different characteristics. Wastewater from crops is characterized by pesticides and chemical products applied to crops (Espigares and Perez 1985). Food processing wastewater has high concentrations of COD, TS, and oil; however, characteristics of food processing wastewater fluctuate substantially depending on what food is processed. For example, COD concentrations of 660, 1,500, 7,400, 800, and 18,000 mg/L are typical for beverage production, breweries, oil and fat production, milk and dairy production, and wheat starch production, respectively (Mori 2003). Slaughterhouse wastewater consists of fats, proteins, fibers, organic carbon, pathogens, and veterinary pharmaceuticals. Typical concentrations are 3,000 mg COD/L, 3,000 mg TS/L, 450 mg TN/L, and 50 mg TP/L (Bustillo-Lecompte and Mehrvar 2017). Livestock wastewater is collected water used for removal of manure from confined spaces where animals are kept, such as milking parlor wastewater. This wastewater is characterized by high organic content and solids, pathogens, and high nutrient contents. For example, Lansing, Víquez et al. (2008) reported 5,720 and 2,330 mg COD/L; 4,570 and 2,180 mg TS/L; 218 and 231 mg TKN/L; and 30.0 and 30.7 mg TP/L, in the wastewater from dairy and swine production, respectively. In the case of anaerobic digestion, livestock wastewater is treated as a semi-solid feedstock. The digestate from anaerobic digesters is then considered to be a special wastewater.

Wastewater treatment aims to reduce the level of pollutants in wastewater prior to reuse or disposal, thereby alleviating the negative impacts of waste production. In large cities, centralized wastewater treatment facilities are common (UN 2015). A centralized wastewater treatment facility is a large-scale wastewater treatment plant that receives wastewater from domestic, commercial, and industrial activities in urban areas (Anderson and Sheffield 2015). This large-scale system manages physical, chemical, and biological processes through a complex combination of mechanical devices that requires technically skilled manpower for operation and maintenance; thus, it is a capital-intensive technology (Avila, García et al. 2016). Instead, a decentralized wastewater treatment facility is an on-site system that treats wastewater from individual users or small communities (UN 2015). Decentralized wastewater treatment facilities tend to be lower in cost and less sophisticated with regards to operation and maintenance than centralized wastewater treatment facilities.

MINSALUD and MINAET have developed the legal framework in terms of the management of wastewater residues. The General Health law (N° 5395, 1973) and Organic Environmental law (N° 7554, 1995) establish legal aspects and regulations with respect to wastewater residues. The General Water Law (No. 276, 1942) establishes water as a public resource for all Costa Ricans and penalizes anyone who contaminates the water. However, this water law is outdated, and fines correspond to 1942. For example, a person would have to pay 180 to 720 colones (~$0.32 to $1.26 in US dollars) if found guilty of contaminating a body of water. In 2017, a new law, the Integrated Management of Water Resources, was discussed in

Congress and, if approved, will replace the outdated water law from 1942. Additionally, in 1961, the Costa Rican Water and Sewage Institute (AyA) was created as the entity to ensure water for human consumption and collection and treatment of wastewater. The Regulation of Dumping and Reuse of Wastewater law (N° 33601-S, 2007) replaced the previous 1997 version (N° 26042-S) and establishes discharge standards for treated wastewater. This wastewater law obligates all industries that generate special wastewater to implement wastewater treatment prior to discharging wastewater into any water body. An operational report of the treatment plant in each industry is requested and if the effluent does not comply standard limits, the MINSALUD can suspend the economic activity and apply fines based on the Regulation Environmental Tax for Dumping (N° 31176, 2003) and the General Water law (Table 2.4).

Table 2. 4. Laws and regulations for the management of liquid residues in Costa Rica.

Law	**Main goal**	**Legal aspects with respect to wastewater**
General Water Law (No. 276, 1942)	To protect and manage water as a public resource in Costa Rica.	Penalize anyone who discharges contaminants into ground and surface water in Costa Rica.
Constitutive Law of the Costa Rican Water and Sewage Institute (N° 2726, 1961)	To establish policies and norms, support and develop programs and plans, and treat water for human consumption and wastewater treatment to assure public health and avoid environmental degradation.	This law created the Costa Rican National Institute of Aqueducts and Sewers (AyA) as the national entity for assuring water for human consumption and wastewater treatment.
General Health law (N° 5395, 1973)	To ensure the public health of the population.	Forbid pollution of ground and surface water due to direct or indirect discharge of solid, liquid and gaseous wastes. Forbid the discharge of industrial residues into the sewage system or storm sewer system. Industries should provide treatment of residues to avoid environmental pollution.
Organic Environmental law (N° 7554, 1995)	To ensure proper management of natural resources. To ensure environmental protection.	Promote plans for protection of the environment. This section includes the implementation of wastewater treatment systems. Industries must provide treatment of residues to avoid environmental pollution.
Regulation Environmental Tax for Dumping (N° 31176, 2003)	To regulate and tax the discharge or spills of residues into water bodies.	Industries must pay a tax based on the quantity of kilograms of COD or TSS discharged into water bodies. The tax must be paid if the concentration of the contaminant (e.g., kg COD/L or kg TSS/L) is greater than the maximum limits for discharge of treated water.
Regulation of Dumping and Reuse of Wastewater law (N° 33601-S, 2007)	To protect public health and the environment by management and treatment of wastewater.	Industries must treat the wastewater. Effluent from wastewater treatment need to comply a discharge standard (e.g., COD < 150 mg/L, TS < 50 mg/L, TN < 50 mg/L, TP < 8 mg/L, $5 < pH < 9$, 15 °C $< T <$ 40 °C, and others). Industries can be suspended and taxed if effluents do not comply with discharge standards (Regulation Environmental Tax for Dumping (N° 31176, 2003)).

In Costa Rica, the legal framework for wastewater seems powerful. Unfortunately, regulations and laws are not enforced, and Costa Rica suffers severe water pollution problems. The clearest example is the AyA, the governmental entity in charge of collection and treatment of ordinary wastewater. For years, multiple sewage systems have collected ordinary wastewater in the central valley; however, the collected wastewater has been simply discharged into the Torres, Rivera, and Maria Aguilar rivers (in the province of San Jose), the Pirro, Burio, and

Bermudez rivers in the province of Heredia, the Zopilote river in Cartago, and El Barro river in Alajuela (Araya, Araya et al. 2003, Angulo 2013). Recently, in 2015, Los Tajos wastewater treatment plant began receiving ordinary wastewater from the approximately 1,700,000 inhabitants of the province of San Jose; however, this wastewater treatment plant only has primary treatment and the effluent complies with discharge standards only rarely (Otarola 2015, Nogarin 2017). Secondary and tertiary treatment units are being planned; however, currently, the effluent from Los Tajos is simply discharged into Torres river (Ruiz 2014, Nogarin 2017).

Treatment of agro-industrial and agricultural wastewater is also extremely limited in Costa Rica. As mentioned previously, only 5,028 industries are registered and only 30% of them presents the operational wastewater treatment report (Angulo 2013). Until responsible execution of the legal framework in Costa Rica occurs, the best approach for the appropriate management of wastewaters is environmental education (Angulo 2013, Calvo 2014). Many efforts have been performed to create decentralized wastewater treatment system within industries, particularly for those in rural areas, where access to sewage systems is unpractical. Alternative treatment systems, also called non-conventional treatment systems, have been proposed by academia and non-governmental organizations. These systems tend to be lower in cost and less sophisticated in operation and maintenance than septic tanks, Imhoff tanks, up-flow anaerobic sludge bed reactors, and activated sludge treatment systems. Biogardens or constructed treatment wetlands (CTWs) have been implemented as alternative treatment systems in Costa Rica (Angulo 2013). The Central American Association for the Economy, Health and the Environment (ACEPESA) has built more than 100 biogardens to treat grey water at houses and hotels in rural communities. In general, this approach seems to be promising for ordinary wastewater in rural areas, where area constraints are not a problem. However, these two approaches have also be applied in urban areas (Moncada 2011, Alfaro, Perez et al. 2013) (Table 2.5). Biogardens and treatment wetlands differ by water content, where biogardens are typically unsaturated and treatment wetlands are continuously saturated.

Table 2. 5. Summary of biogarden and constructed treatment wetland projects developed by academic institutions.

Project	**Institution (Reference)**	**Wastewater type**	**Treatment system**	**Dimensions (Length x width x height, m)**	**Water quality parameter (In / Out, mg/L)**			
					COD	**TS**	**TN**	**TP**
Barra Honda, Guanacaste	UNA / ACEPESA (Cubillo and Gomez 2017)	Grey water	Biogarden	12 x 4 x n.r.	161 / 132	n.r.	0.16 / 0.029 [1]	0.137 / 0.246
Barra Honda, Guanacaste	UNA / ACEPESA (Cubillo and Gomez 2017)	Grey water	Biogarden	12 x 4 x n.r.	54.0 / 13.0	n.r.	0.089 / 0.116 [1]	0.088 / 0.05
Barra Honda, Guanacaste	UNA / ACEPESA (Cubillo and Gomez 2017)	Grey water	Biogarden	12 x 4 x n.r.	140 / 75.0	n.r.	0.024 / 0.03 [1]	0.07 / 0.066
La Virgen, Guanacaste	UNA / ACEPESA (Cubillo and Gomez 2017)	Grey water	Biogarden	12 x 4 x n.r.	145 / 22.0	n.r.	0.022 / 0.107 [1]	0.043 / 0.22
La Virgen, Guanacaste	UNA / ACEPESA (Cubillo and Gomez 2017)	Grey water	Biogarden	12 x 4 x n.r.	54.0 / 26.0	n.r.	0.03 / 0.082 [1]	0.05 / 0.046
Zapote, San Jose	TEC / ACEPESA (Moncada 2011)	Grey water	Biogarden	5 x 1 x 0.7	444 / 62	60 / 6.2	4.5 / 6.7	3.5 / 2.5
n.r.	UNA (Pérez, Alfaro et al. 2013)	Black water from a septic tank	HSSF-CTW	12 x 3 x 0.6	250 / 50	400 / 108	n.r.	8 / 3
Popular Cultural Museum, Heredia	UNA (Alfaro, Perez et al. 2013)	Grey water	HSSF-CTW	8 x 2.5 x 0.7	98% [2]	99% [2]	n.r.	n.r.
Monteverde, Puntarenas	(Dallas, Scheffe et al. 2004)	Grey water	HSSF-CTW	12 x 1.2 x 0.5	167 / 10 3	15 / 6	8.4 / 1.1 [4]	1.6 / 3.6

n.r.: not reported. HSSF-CTW: horizontal subsurface flow constructed treatment wetland

[1] reported as ammonium.

[2] only treatment performance was reported.

[3] reported as BOD.

[4] reported as ammonium.

The major limitation in terms of wastewater treatment in Costa Rica is the lack of compliance and enforcement with the current legal framework. Efforts conducted by academic institutions increase the public's awareness of water pollution problems in Costa Rica. Environmental education, until now, has been the best tool for improving sanitation and decreasing water pollution in Costa Rica. However, more research is needed on design of biogardens and CTWs. The projects reported in Table 2.5 focus on treatment performance under specific conditions; general design parameters were not established. Thus, design of biogardens and CTWs in Costa Rica continues to be empirical. Instead, to promote development of CTW, more engineering analyses are needed.

2.2.3. Biomass residues and wastewater: the problem becoming an opportunity

In part due to improper management of biomass residues and wastewater, Costa Rica has a severe water pollution problem. A simple indicator of this problem is poor water conditions in most surface waters in Costa Rica. For example, the Regional Institute for Studies on Toxic Substances (IRET) sampled 487 points in 250 rivers across Costa Rica, and found that 71% of the sampling points have a moderate to extreme pollution condition (IRET 2012). Laws and regulations exist; however, the legal framework is not enforced or properly followed by the regulators. Therefore, proper management of wastewater depends on social will and awareness of agro-industry and agricultural activities. However, this contamination problem can become an opportunity for much needed change. Environmental education and academic efforts can turn the current linear economic models into circular, close-loop, economic models. Integrating decentralized self-sufficient, close-loop, organic waste treatment systems into current agro-industry and agricultural activities is expected to promote a circular economy that considers residues as resources, alleviating the demand of resources and reducing waste generation and disposal (Figure 2.1). Additionally, Costa Rica has two important related goals, 1) to be the first carbon neutral country by 2021 and 2) to develop a green economy. Thus, integration of treatment systems into current agro-industry and agricultural activities can solve the water pollution problem, while helping Costa Rica achieve carbon neutrality and green development goals (Table 2.6).

Table 2. 6. Biomass residues and wastewater: the problem and subsequent opportunities.

<table>
<tr><th>Costa Rican goals</th><th>Action taken by Costa Rica</th><th>The problem</th><th>The opportunity</th><th>Benefits</th></tr>
<tr><td>Carbon neutrality by 2021</td><td>Generation of energy using renewable resources.</td><td>Current energy production from biomass is only 5.44% of the potential energy.</td><td rowspan="2">Integrate decentralized self-sufficient, close-loop, organic waste treatment system to consider residues as resources.</td><td rowspan="2">Energy generation from biomass resources.
Reduction of fossil fuel use.
Reduction of resource consumption and waste disposal.
Recovery of nutrients.
Water reclamation.
Reduction of sanitation problems.</td></tr>
<tr><td>Green development</td><td>Optimization of resource utilization.</td><td>Current linear economic model does consider residues.
Biomass residues and wastewater are not properly treated.</td></tr>
</table>

2.3. Integration of technologies at the tropics: Costa Rica case

A discussion of technologies for treatment of biomass residues and wastewater is presented here. Integration of technologies can overcome individual disadvantages of each technology. The main goal of the SPAD-HCTW is implementation of a decentralized self-sufficient, close-loop, organic waste treatment system. The integrated system should provide energy, fertilizers, and reclaimed water, to contribute to the protection of water resources in Costa Rica. In addition, this approach can reduce the demand of water, energy, and food resources (Figure 2.1). This integration can contribute to sustainable development in rural areas in Costa Rica, in particular with regards to agro-industry and agricultural activities.

2.3.1. Waste to energy: optimizing anaerobic digestion through solar thermal collection

Anaerobic digestion is a solid waste management technology for processing high moisture content biomass residues into energy (IRENA 2012, Funk, Milford et al. 2013). Anaerobic digestion starts with hydrolysis of lipids, complex polymers (e.g. cellulose, polysaccharides), and particulate organic materials into monomers (e.g., sugar and amino acids) and long chain fatty acids. Then, in acidogenesis, monomers are fermented or anaerobically oxidized into short chain fatty acids, alcohol, and ammonia. In acetogenesis, alcohol and short chain fatty acids are converted to acetic acid, hydrogen (H_2) and/or carbon dioxide (CO_2). Finally, methanogens, in the absence of oxygen, produce biogas from acetic acid, CO_2, and H_2 (Gould and Crook 2010). Solid and liquid digestate are also products of anaerobic digestion. Solid digestate can be used as a fertilizer to enhance nutrient retention in soils (Liedl, Bombardiere et al. 2006). Then, to prevent environmental impairment and to allow reuse of the water, liquid digestate can be treated. The treatment of the liquid digestate in discussed further.

Biogas is primarily a mixture of methane (CH_4) and CO_2. Small amounts of hydrogen sulfide (H_2S), water vapor (H_2O), nitrogen (N_2), H_2, and oxygen (O_2) are minor components in biogas. Methane is a relatively dense and reliable biochemical energy source (Tsalkatidou, Gratziou et al. 2009). The simplest utilization of CH_4 as energy is use of biogas as a cooking fuel (Kinyua 2015). Similarly, biogas can be burnt for heating fluids (e.g., water, air, and even, digestate) or buildings. Finally, CH_4 can be used to fuel electrical generators, such as combined heat and power (CHP) systems. Importantly, before using CH_4, it is recommended to remove any H_2S from the biogas to prevent formation of sulfuric acid, a corrosive liquid. Iron steel wool can be placed inside biogas effluent pipes for simple use of the biogas, such as a cooking fuel; activated carbon filters can also be used to remove H_2S from biogas prior to CHP (Kinyua 2015, Noramelya, Shahbudin et al. 2016).

In Costa Rica, the most common anaerobic digester reactors are covered lagoon and tubular polyethylene bag digesters (Table 2.3). Covered lagoon and tubular polyethylene bag digesters are attractive to inhabitants in rural areas due to simplicity of installation and operation, as well as low investment costs and energy savings (Lansing, Víquez et al. 2008). However, due to the simplicity of these systems, anaerobic digestion performance can be affected by lack of control over operating temperatures, instability of operating temperatures, and limited mixing. Due to warmer ambient temperatures in the tropics (15 – 30°C), covered lagoons and tubular polyethylene bag digesters are not typically heated. Thus, these digesters work at operating temperatures close to the low mesophilic range. For example, at Earth University, Costa Rica, Lansing, Víquez et al. (2008) reported temperatures of 26.7 and 25.8°C from a 85 m^3 tubular polyethylene bag digesters receiving cow and swine manure, respectively. Consequently, microbial growth and organic matter degradation is lower than that reported for higher thermophilic operating temperatures. For example, Vindis, Mursec et al. (2009) used 0.5 L mini digesters fed with maize to demonstrate that higher biogas production (494 – 611 NL kg/VS) was achieved at 55°C, as compared to 315 – 409 NL kg/VS produced at 35°C. In addition, as the operating temperature depends on the ambient temperature, changes in ambient temperature can produce instability in the operating temperature. For example, Visser, Gao et al. (1993) indicated that instability of temperature decreased microbial growth and CH_4 production. In fact, Espinosa-Solares, Valle-Guadarrama et al. (2009) observed that methane production decreased 11.1% per °C for small changes in temperature from 52 to 56°C. Moreover, contents in covered lagoons and tubular polyethylene bag digesters are not mixed. Without mixing, localized pockets of temperature can create heterogeneous operating temperatures due to stratification of solids and formation of dead zones (Espinosa-Solares, Valle-Guadarrama et al. 2009, Suryawanshi, Chaudhari et al. 2010). In contrast, through mixing, Bombardieri, Espinosa-Solares et al. (2007) maintained a 40 m^3 continuous stirred tank reactor (CSTR) anaerobic digester temperature at stable ±0.1°C from the target temperature, allowing steady-state conditions for biogas production. Mixing also allows homogeneity of pH, better interaction between microbes and substrate, and release of generated biogas (Suryawanshi, Chaudhari et al. 2010). In addition, temperatures close to the psychrophilic (<25°C) and mesophilic (25-40°C) range do not assure pathogenic destruction. For example, Kinyua (2015) detected pathogens (e.g., *Giardia lamblia* and *Cryptosporidium parvum*) in the effluent of a swine tubular polyethylene bag digester working at 20.7°C. Instead, at thermophilic (<45°C) conditions, there is thermal destruction of pathogens (Vindis, Mursec et al. 2009).

Therefore, to improve utilization of energy in biomass residues and eliminate pathogens in the digestate, a thermophilic CSTR anaerobic digester is recommended. Thermophilic anaerobic digestion speeds up microbial growth and organic matter degradation, consequently producing more biogas and discharging pathogen free digestate. A CSTR is characterized by continuous feeding of feedstock and discharge of digestate, and continuous mixing inside the digester. Therefore, the CSTR reactor type assures homogeneity of parameters (e.g., temperature, chemical concentration, pH, alkalinity, and substrate concentration) inside the digester, promoting steady-state conditions for biogas production (Bharati and Kalamdhad 2016). In addition, numerous studies have demonstrated that thermophilic cultivation can enhance anaerobic digestion performance by shortening retention time, and thus requiring smaller vessel sizes, improving odor control, and reducing total solids in waste streams (Aitken, Sobsey et al. 2005, Suryawanshi, Chaudhari et al. 2010, Sharma, Espinosa-Solares et al. 2013, Zarkadas, Sofikiti et al. 2015).

However, as a disadvantage, a thermophilic CSTR anaerobic digester requires higher input energy compared to covered lagoon and tubular polyethylene bag digesters working at ambient temperature. Energy is required for mixing and maintaining thermophilic temperature. For conducting mixing, energy input is required to operate a blower for recirculating the biogas or a pump for recirculating the digestate. For example, in a 40 m^3 CSTR anaerobic digester, Espinosa-Solares, Valle-Guadarrama et al. (2009) reported that a blower and a digester recirculation pump consumed per day 3.18% and 1.28% of the total energy needed to operate the digester (428 MJ). The major input energy is heat for maintaining thermophilic conditions; 95.5% of the input energy was required to heat the CSTR anaerobic digester (Espinosa-Solares, Valle-Guadarrama et al. 2009). Thus, heating represents the major disadvantage to thermophilic digesters.

One strategy for heating thermophilic CSTR anaerobic digesters is using the biogas generated. However, heating using the biogas may lead to an unfavorable energy balance for small-scale digesters (Vindis, Mursec et al. 2009). In contrast, higher biogas production in large-scale reactors covers heating demand to maintain thermophilic temperature and provides surplus energy for electric generation. For example, Zabranska, Dohanyos et al. (2002) demonstrated that a 4,800 m^3 thermophilic (55°C) CSTR anaerobic digester treating raw sludge used 167 MWh/d for heating the system and produced 108 MWh/d as a surplus energy. Thus, the biogas produced was mainly used for heating purposes (Zabranska, Dohanyos et al. 2002).

To overcome the higher energy requirement of thermophilic anaerobic digestion, solar heating systems have been implemented. Low-density and inconsistent solar energy can be converted into a relatively dense and reliable biochemical energy source – methane. In Jordan, Alkhamis, El-khazali et al. (2000) installed a 1.54 m^2 flat-plate solar thermal collector to heat a laboratory scale reactor (volume = 0.053 m^3). Hot water from the collectors was stored in a storage tank (volume = 0.096 m^3). A heat exchanger passing through the storage tank and the reactor recirculated water to maintain a constant temperature of 40°C in the digester (Figure 2.2.a). In Greece, Axaopoulos, Panagakis et al. (2001) and Axaopoulos and Panagakis (2003) installed a 21 m^2 flat-plate solar thermal collectors to heat an underground anaerobic digester (total volume 116 m^3, with a useful volume of 40 m^3). Hot water was pumped through a heat exchanger installed on the bottom of the anaerobic digester to maintain a digestion temperature of 35°C. In 2001, the authors developed a mathematical model to describe the dynamic behavior of the system, which showed a good agreement with the measured values. Then, in 2003, results showed sufficient biogas production from the digester to heat a swine nursery during winter (Figure 2.2.b). In Egypt, El-Mashad, van Loon et al. (2004) targeted a thermophilic (50°C) process in a 10 m^3 completely stirred tank reactor (CSTR) anaerobic digester. The authors evaluated two different configurations, one with the solar collector separate from the digesters (Figure 2.2.c) and the other with the solar collector integrated into the digester (Figure 2.2.d). The area covered by the solar collector was 4.88 m^2. Solar-heated water maintained the thermophilic temperature most of a year except during cold months.

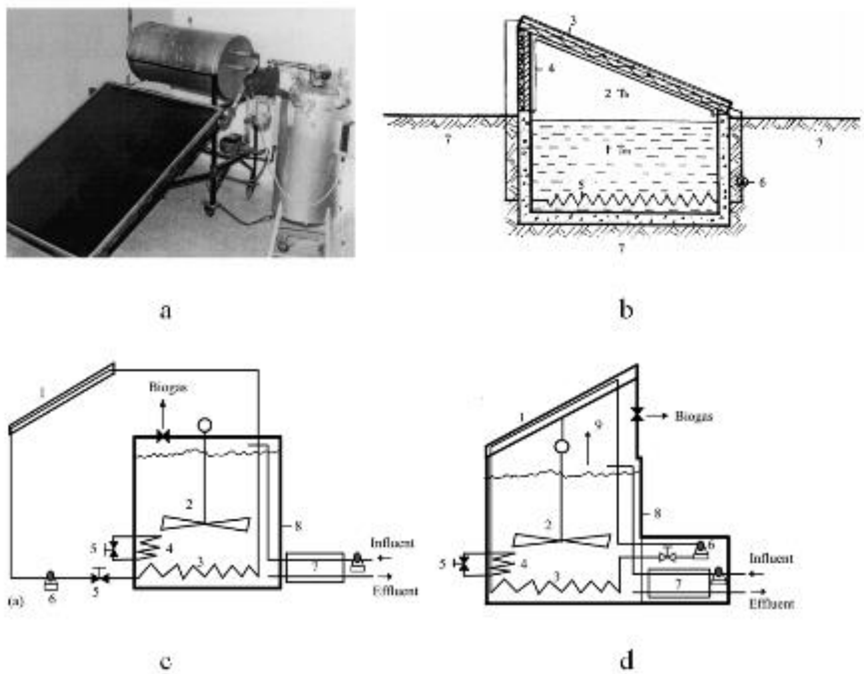


Figure 2. 2. A schematic of the system configuration of integrated solar thermal collectors and anaerobic digesters. a) Alkhamis, El-khazali et al. (2000); b) Axaopoulos, Panagakis et al. (2001) and Axaopoulos and Panagakis (2003); and c) and d) El-Mashad, van Loon et al. (2004).

These studies have shown that solar thermal collectors can supply the energy required to maintain a specific temperature during digestion. However, during winter in the temperate regions (e.g., Egypt) a portion of biogas produced is needed to heat thermophilic anaerobic digesters (El-Mashad, van Loon et al. 2004). Instead, in the tropics, it is expected that solar thermal collectors can maintain thermophilic conditions due to warmer ambient temperature (15 – 30°C). However, even with constant, higher temperatures, effluent digestate from thermophilic digesters still has high solid, carbon, and nutrient contents, and improper disposal can cause water pollution.

2.3.2. Constructed treatment wetlands: ending the water pollution problem

Natural wetlands are ecosystems that have these attributes: 1) saturated soil or soil covered by shallow water, 2) hydric soils, or 3) presence of macrophytes (Mitsch and Gosselink 2007). Therefore, a CTW is an engineered modification of the landscape to mimic a natural wetland ecosystem. This man-made system is intended to create an ecosystem that stabilizes, sequesters, accumulates, degrades, metabolizes, and mineralizes nutrients in the wastewater as natural wetlands do (Halverson 2004, Vymazal 2007). The longevity of CTWs varies depending on the system; however, estimations vary from 10 to 15 years (Wallace and Knight 2006). As mentioned previously, CTWs are a decentralized wastewater treatment technology appropriate for individual users or small communities in rural areas that tend to be lower in cost and less sophisticated in operation and maintenance than centralized wastewater treatment facilities (Table 2.5). In addition, CTWs have been implemented for treating industrial and agricultural wastewater (Vymazal 2014) and high-strength wastewaters (Paing, Serdobbel et al. 2015).

Based on the hydrologic criteria, CTWs can be categorized as horizontal subsurface flow (HSSF), vertical subsurface flow (VSSF), and free water surface (FWS) wetlands. Hybrid constructed treatment wetlands (HCTW) consist of, in any sequence, a combination of HSSF, VSSF, and FWS wetlands to achieve specific treatment goals.

2.3.2.1. Horizontal subsurface flow wetlands

HSSF wetlands consist of a filter media planted with emergent plants. Filter media, usually sand or gravel, and roots provide surface area for microorganisms to grow. Roots exude oxygen into the rhizosphere to support aerobic microorganism metabolism. Wastewater is kept below the surface and flows, from the inlet to the outlet, horizontally through the filter media and in and around the roots (Kadlec and Wallace 2009). The wastewater is kept at a constant level in the filter media and continuous saturation promotes anoxic/anaerobic conditions due limited oxygen transfer capacity. Thus, anoxic conditions prevail in HSSF wetlands and facilitate chemical reduction processes (Vymazal 2007, Kadlec and Wallace 2009). Limited aerobic zones occur around roots and rhizomes; thus aerobic chemical oxidation processes in HSSF wetlands are limited. Because wastewater is not exposed to the atmosphere, HSSF wetlands do not provide habitat suitable for mosquitoes, provide odor control, and minimize exposure to pathogenic organisms (Vymazal 2007, Kadlec and Wallace 2009, Stefanakis and Tsihrintzis 2012) (Figure 2.3).

HSSF wetlands have demonstrated to be effective at removing organic matter and suspended solids. Organic matter can be removed through biological degradation by aerobic and anaerobic microbial processes and microbial uptake (e.g., organic matter serves as carbon sources and energy for nitrification and denitrification). Sedimentation and filtration remove suspended solids. For example, Summerfelt, Adler et al. (1999) applied 30 kg/m^2/yr at 0.6 m^3/d of sludge from an aquaculture system into a 4.5-m^2 HSSF wetland. The removal of organic matter and suspended solids were 97% and 74% from an inlet concentration of 6,855 mg COD/L and 7,860 TSS/L, respectively (Summerfelt, Adler et al. 1999). Clogging is potential concern during operation of HSSF wetlands. The inlet area can clog with organic matter and suspended sediment filtered and trapped in the media. For example, De Paoli and Sperling (2013), reported clogging and surface runoff in a HSSF wetland planted with *Typha latofolia* as the inlet area accumulated 35 g VS/L.

The main process for nitrogen removal in wetlands is nitrification followed by denitrification. Nitrification is restricted due to limited aerobic zones in HSSF wetlands.

Nitrification is a two-step process driven by strictly aerobic nitrifying bacteria. First, *Nitrosomas* oxidize the ammonium (NH_4) to nitrite (NO_2^-), and then, *Nitrobacter* oxidize NO_2^- to nitrate NO_3^-. Both sequential processes uses oxygen as an electron acceptor, carbon dioxide as a carbon source, and either ammonium or nitrite as source of energy (Vymazal 2007) (equations 2.1). Instead, anaerobic conditions prevail in the HSSF wetland and the NO_3^- is reduced to nitrogen gas (N_2) by denitrification. For denitrification, organic matter is used as a carbon source by denitrifiers, while nitrate is used as electron acceptor (Vymazal 2007) (equation 2.2). For example, for a 54-m^2 HSSF wetland treating tilapia production wastewater, Zachritz, Hanson et al. (2008) reported 76% removal of NO_3^-, whereas removal of NH_4 was low (7.5%). Zachritz, Hanson et al. (2008) indicated that the HSSF wetland was oxygen limited. Carbon can also be a limiting factor for denitrification. For example, only 13% of NO_3^- was removed from a 10-m^2 HSSF wetland receiving effluent from a VSSF wetland (Soroko 2007). Soroko (2007) indicated that carbon was the limiting factor because the VSSF wetland removed from 2,448 mg BOD/L from 2,452 mg BOD/L in the wastewater. Minor mechanisms that also remove nitrogen from wastewater are ammonia volatilization, ammonia adsorption, and plant uptake if plants are harvested (Vymazal 2007).

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad \text{[Equation 2.1]}$$

$$6(CH_2O) + 4NO_3^- \rightarrow 6CO_2 + 2N_2 + 6H_2O \qquad \text{[Equation 2.2]}$$

In HSSF wetlands, phosphorus (P) is removed by plant uptake, microbial uptake, sorption, and precipitation. Phosphorus removal by plant uptake only occurs if vegetation is harvested, otherwise, when plants decays, phosphorus is released back into the water. Microbial uptake by bacteria, fungi, algae, and microinvertebrates occurs quickly but phosphorus is released when the microorganisms decay. Sorption depends on the material of the filter media.

For example, sand has more sorption capacity than gravel or crushed rock. Minerals such as reactive iron, aluminum hydroxide, or oxide groups on the surface area of filter media increase adsorption capacity. In addition, calcareous materials can promote phosphorus precipitation. However, filter sorption sites can become saturated, limiting phosphorus sorption. Thus, phosphorus moves in a sedimentary cycle, and phosphorus removal from wetlands tends to be low (Vymazal 2005, Kadlec and Wallace 2009, Vymazal 2014). For example, Avila, Salas et al. (2013) reported that a 229-m^2 HSSF receiving effluent from a VSSF wetland treating sewage wastewater only removed 22% of influent phosphorus. In fact, further experiments with the same system indicated no retention of P as the effluent concentration remained equal or more than the inlet concentration to the HSSF wetland (Avila, García et al. 2016).

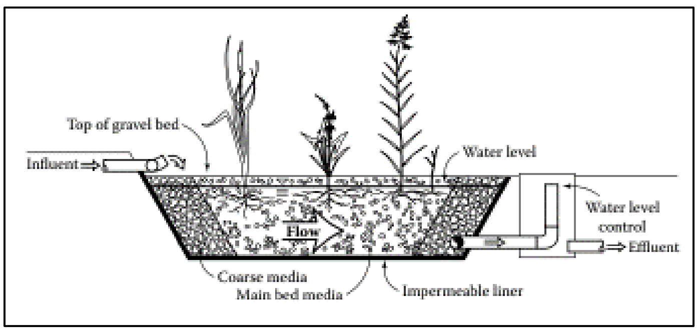


Figure 2. 3. A schematic of a HSSF wetland. Figure was taken from Kadlec and Wallace (2009).

2.3.2.2. Vertical subsurface flow wetlands

VSSF wetlands have a similar configuration to HSSF wetlands, with the exception of the direction of wastewater flow. In VSSF wetlands, wastewater can flow downward or upward and either intermittently or continuously. In downward-flow wetlands, the wastewater moves from the top of the surface media to the bottom; whereas, in upward-flow systems, wastewater fills from bottom to top. In tidal flow wetlands (a specific type of upward-flow wetland), the wastewater is applied from the bottom, moving upward until filling the substrate, and then the wastewater is drained (Kadlec and Wallace 2009). During intermittent downward-flow application, wastewater floods wetland surface, then moves downward through the filter media by gravity. The intermittent application of the wastewater allows for higher levels of oxygen transfer. Consequently aerobic conditions prevail in the filter media and facilitates chemical oxidation processes (Vymazal 2007). In VSSF wetlands, the wastewater is only momentarily exposed to the atmosphere, thus VSSF wetlands do not provide habitat suitable for mosquitoes and partially provide odor control and minimize exposure to pathogenic organisms (Vymazal 2007, Kadlec and Wallace 2009, Stefanakis and Tsihrintzis 2012) (Figure 5.4).

In VSSF and HSSF wetlands, similar processes account for removal of organic matter and suspended solids. However, removal is greater due to aerobic conditions in VSSF wetlands. For example, Summerfelt, Adler et al. (1999) treated sludge from the tilapia aquaculture system (0.6 m^3/d at 30 kg/m^2/yr) using a 4.5 m^2 VSSF wetland, and, compared to the HSSF wetland, the authors found superior removals of organic matter (92%) and suspended solids (98%) from an inlet concentration of 6,855 mg COD/L and 7,860 TSS/L, respectively (Summerfelt, Adler et al. 1999). Filtered and trapped organic matter and suspended sediments can impact the filter media of VSSF wetlands; however, intermittent application of wastewater minimize clogging due to degradation of organic matter (i.e., volatile solids) during the resting period (Leverenz, Tchobanoglous et al. 2009).

In VSSF wetlands, the primary pathway for nitrogen removal is nitrification followed by denitrification. Aerated conditions in the filter media prevail, thus VSSF wetlands have the ability to oxidize NH_4 to NO_3^- by nitrification (equation 2.1). For example, in a laboratory scale VSSF wetland (surface area = 0.13 m^2), Xinshan, Qin et al. (2010) treated high concentration nitrogenous domestic wastewater. Due to high dissolved oxygen (DO) concentration (>1.5 mg/L) the NH_4 was mostly converted to NO_3^- in a hydraulic retention time (HRT) of two days; however, extended HRT reduced nitrification rates due to lack of carbon source (Xinshan, Qin et al. 2010). In contrast, in VSSF wetlands, denitrification is limited (equation 2.2). This is reported in Summerfelt, Adler et al. (1999) by data showing 45 mg NO_3^-/L from the VSSF wetland effluent compared to 0.38 mg NO_3^-/L from the HSSF wetland effluent. Likewise, minor mechanisms, such as ammonia volatilization, ammonia adsorption, and microbial and plant uptake, enhance TN removal from the wastewater (Vymazal 2007). For example, NO_2^- and NO_3^- can be assimilated by microbial and plant biomass. However, nutrients can be released back if microbes or plants die in the wetland. Thus, total nitrogen removal by plants only occurs if plants are harvested (Kadlec and Wallace 2009, Avila, Salas et al. 2013).

Plant uptake, microbial uptake, sorption, and precipitation are responsible for phosphorus removal in VSSF CTW; however, removal is limited (Vymazal 2007, Stefanakis, Akratos et al. 2014, Vymazal 2014). Phosphorus is removed by plant uptake only if plants are harvested. Microbes uptake phosphorus, which is released back when microbes die. Similar to HSSF wetlands, sorption is typically the primary removal mechanisms but depends on the material of the filter media; additionally, phosphorus sorption is limited because of the relatively rapid movement of wastewater through the filter media (Stefanakis and Tsihrintzis 2012). Compared to a downward-flow VSSF wetland, sorption of phosphorus was higher in an upward-flow VSSF wetland due to longer HRT (Campbell and Safferman 2015). Similarly, in HSSF wetlands, removal capacity of the filter media decreases with time. For example, Campbell and Safferman (2015) reported that phosphorus removal decreased from 74% at the beginning of the experiment to 30% at the end of the experiment due to less availability capacity in the filter media. Similarly, after 3 years of operation of a VSSF wetland, no phosphorus removal was reported by Avila, García et al. (2016) as effluent concentrations were equal to inlet concentrations.

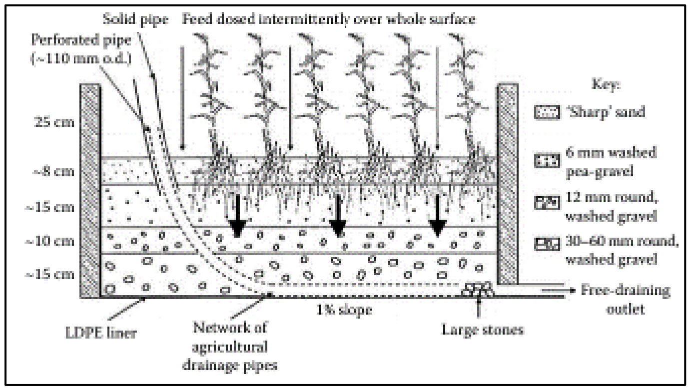


Figure 2. 4. A schematic of a VSSF wetland. Figure was taken from Kadlec and Wallace (2009).

2.3.2.3. Free water surface wetland

FWS wetlands consist of an area of open water, usually a shallow channel or basin, in which the wastewater freely flows above the ground surface (Kadlec and Wallace 2009, Stefanakis and Tsihrintzis 2012). Different types of plants can grow in FWS wetlands. Emergent plants are rooted in the soil growing beneath the water surface, with leaves, stems, and reproductive parts above the water, while submerged plants are also rooted in the soil, with all their parts growing beneath the water surface (Mitsch and Gosselink 2007). Free floating plants have their roots hanging freely in the water, with leaves and stems floating on the water surface; whereas, floating-leaved plants have leaves that float on the water surface but roots that are grounded in the soil (Mitsch and Gosselink 2007). Parts of plants beneath the water surface (roots, leaves, stems) and roots in the underlying sediments provide surface area for microorganisms to grow. Roots exude oxygen into the surrounding areas for aerobic microbial activities. In addition, via photosynthesis, algae supply oxygen in the water column; however, algae activity can decrease if floating plants cover the wetland surface. Because standing wastewater is exposed to the atmosphere, FWS wetlands provide habitat suitable for mosquitoes and there is a health risk due to exposure of pathogenic organisms (Vymazal 2007, Kadlec and Wallace 2009). Thus, to avoid any health risks, FWS are used for secondary or tertiary treatment. In addition, due to the open water area, the DO concentration varies from high (near the surface) to low (near the bottom) (Stefanakis, Akratos et al. 2014). Thus, a variety of chemical oxidation and reduction processes can occur within the water column (Figure 2.5).

FWS wetlands can be effective at removing suspended solids and organic matter. Sedimentation, filtration, aggregation, and surface adhesion can remove suspended sediments. Particles settle and filter out as the wastewater flows through the wetland vegetation. Vegetation enhances sedimentation by reducing water velocities, intercepts, and filters particles so that the particles are sorbed to surface areas of the plants. El-Sheikh, Saleh et al. (2010) reported that suspended solids were reduced by 76% by a 12,500 m^2 FWS wetland treating 4,300 m^3/d of wastewater from the Bahr El Baqar drain in Egypt. Like suspended sediment removal, organic matter can be removed through deposition and filtration, particularly the largest and heaviest particles. In addition, biological degradation removes organic matter. Aerobic degradation can be performed by microbial growth in the water column, whereas anaerobic degradation occurs mainly at the bottom of the wetland. Burgoon, Kadlec et al. (1999) reported 87% of COD removal for a 100,000-m^2 FWS wetland receiving potato processing wastewater with 2,528 mg COD/L. Removal was in part performed by a large population of aerobic and anaerobic microorganisms, and facilitated by larger wetland size and HRT (Burgoon, Kadlec et al. 1999, El-Sheikh, Saleh et al. 2010).

Nitrification and denitrification is the most effective process for nitrogen removal in FWS wetlands. As mentioned before, the FWS wetlands provide aerobic zones (water column) to oxidize NH_4^+ to NO_3^- and anoxic zones (sediments) to reduce NO_3^- to nitrogen gas or nitrous oxide. For example, Kapellakis, Paranychianakis et al. (2012) reported high removal efficiencies of NH_4^+ and NO_3^- due to paired nitrification and denitrification in a FWS wetland treating olive mill wastewater. Minor mechanisms that also remove nitrogen from wastewater are ammonia volatilization, ammonia adsorption, anaerobic ammonium oxidation (ANAMOX), organic nitrogen burial, and plant uptake if plants are harvested (Vymazal 2007). In general, FWS provide aerobic and anaerobic environments and carbon for nitrogen removal. Consequently, larger FWS wetlands with larger HRT have demonstrated high removal efficiencies. However, this becomes a disadvantage if land is not available or costly (Burgoon, Kadlec et al. 1999, El-Sheikh, Saleh et al. 2010).

Phosphorus can be removed by plant uptake, microbial uptake, adsorption, and soil formation in FWS wetlands. Plant and microbes incorporate phosphorus in their tissues; however, phosphorus can be released back to the wetland if their organic matrices degrade. By harvesting plants, phosphorus can be physically removed from the wetland. Adsorption of phosphorus occurs in the soil surface and increases in soils with high clay content and high concentrations of aluminum, iron, and calcium. In addition, phosphorus precipitation can also occur if aluminum, iron, and calcium are in the soil. However, phosphorus can be re-dissolved under altered conditions. In fact, FWS wetlands can export phosphorus. Long-term phosphorus removal, which is commonly thought to be limited to ~10% of influent phosphorus, is due to soil formation based on sediments and litter (Vymazal 2007, Kadlec and Wallace 2009).

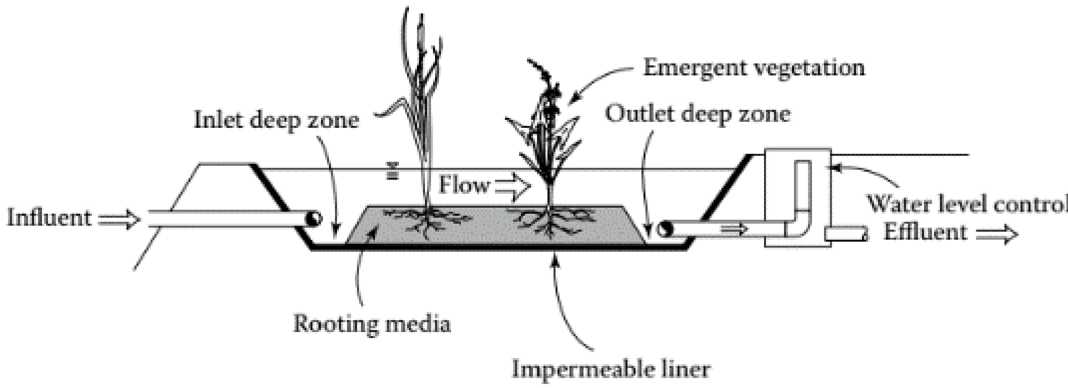


Figure 2. 5. A schematic of a FWS wetland. Picture was taken from Kadlec and Wallace (2009).

2.3.2.4. Hybrid constructed treatment wetland

HCTWs combine multiple types of wetlands to overcome disadvantages of individual treatment wetlands types. Depending on wastewater characteristics, type, quantity, and loading, wetlands can be selected to achieve higher removal efficiencies. The VSSF wetland followed by a HSSF wetland configuration targets NH_4 removal by nitrification in the VSSF wetland followed by denitrification in the HSSF wetland. Carbon source could be limited for denitrification by high removal of organic matter at the VSSF wetland; however, this is of limited concern for treatment of high strength wastewaters. Combining a VSSF wetland with a FWS wetland also promotes sequential nitrification and denitrification. In this case, carbon can be supplied by decomposed organic matter at the bottom of the FWS wetland. A third option for nitrogen removal could be a sequential HSSF wetland with a VSSF wetland for high concentrations of NO_3^- in the wastewater. Denitrification is performed in the HSSF wetlands using organic matter in the wastewater as carbon source. Then, nitrification is performed in the VSSF wetland, and some portion of the effluent can be recirculated into the HSSF for denitrifying, enhancing the TN removal. Table 2.7 demonstrates the efficacy of different HCTW configurations.

Table 2. 7. Hybrid constructed treatment wetland efficacy.

Location	Type of wastewater	Type of CTWs	Water quality parameter (In / Out, mg/L)					
			COD	TSS	NH_4-N	NO_3-N	TN	TP
Denmark (Brix, Arias et al. 2003)	Sewage	HSSF + VSSF	376 / 50 50 / 36	n.r. n. r.	17 / 16 16 / 0.4	2 / 3.1 3.1 / 6.6	40.6 / 20.7 20.7 / 16.8	n.r. n.r.
Nepal (Laber, Haberl et al. 2003)	Sewage	HSSF + VSSF	162 / 45 45 / 10	57 / 19 19 / 1.5	32 / 27 27 / 0.1	0.2 / 0.4 0.4 / 27	n.r. n.r.	4.4 / 2.6 2.6 / 1.4
Spain (Serrano, de la Varga et al. 2011)	Winery	VSSF + HSSF	1,558 / 711 711 / 448	129 / 65 65 / 17	28 / 19 19 / 12	n.r. n.r.	52.9 / 26.0 26.0 / 25.2	n.r. n.r.
Spain (Avila, García et al. 2016)	Sewage	VSSF + HSSF + FWS	335 / 147 147 / 82 82 / 73	166 / 47 47 / 12 12 / 8	34 / 17 17 / 10 10 / 7.2	** / 14 14 / 13 13 / 7.7	n.r. n.r. n.r.	3.2 / 3.2 3.2 / 3.2 3.2 / 3.2

**: below limit of detection
n.r.: no reported

2.3.2.5. Selection criteria

In general, for the same wastewater flow and characteristics, VSSF wetlands achieve higher removals than FWS due to predominant aerobic conditions and high specific surface area of the media. Removal of organic matter, suspended solids, and NH_4 are higher in VSSF wetlands than in FWS wetlands. In contrast, NO_3^- removal is higher in HSSF wetlands. FWS wetlands have also demonstrated high removal efficiencies; however, FWS wetlands require larger areas to achieve the same treatment VSSF or HSSF wetlands. If land is available and inexpensive, FWS wetlands can be an attractive option. Investment, maintenance, and operation costs are lower for FWS wetlands than for VSSF and HSSF wetlands. If land is not available, VSSF and HSSF wetlands can be reliable options. However, both systems are relatively costly compared to a FWS wetland. The selection of subsurface wetland type would depend on the target pollutant. If wastewater has high NH_4 concentration, a VSSF is preferable, but if the target is the removal of NO_3^-, a HSSF is suggested. Table 2.8 summarizes CTW treatment properties for selection criteria.

Table 2. 8. Treatment properties of constructed treatment wetlands.

CTW type	Treatment properties[1]
FWS	Excellent reduction of organic matter and suspended solids. Excellent for denitrification of nitrate and final removal of total nitrogen. Not effective nitrifying ammonium to nitrate.
HSSF	Excellent reduction of organic matter and suspended solids. Excellent for denitrification of nitrate and final removal of total nitrogen. Not effective at nitrifying ammonium to nitrate. Compared to a FWS: For the same wastewater (characteristic and flow rate), the HSSF possesses higher investment cost, but requires less treatment area, provides mosquito control, odor control, and avoid contact with the wastewater.
VSSF	Excellent reduction of organic matter and suspended solids. Excellent for nitrification of ammonium. Not effective denitrifying nitrate to nitrogen. Compared to HSSF: For the same wastewater (characteristic and flow rate), the VSSF requires less treatment area, but require more operation and maintenance. Both wetland types have similar investment cost, provides mosquito control, odor control, and avoid contact with the wastewater.

[1] (Kadlec and Wallace 2009)

Therefore, in Costa Rica, a HCTW can be implemented to treat the wastewater from a thermophilic CSTR anaerobic digester and assure water treatment. The implementation of CTWs has been demonstrated (Table 2.5); however, these systems have been designed for sewage wastewater. The effectiveness and removal capacity of a CTW treating the effluent from a thermophilic CSTR anaerobic digester in Costa Rica has not been reported. The design of CTWs, including the evaluation of the hydrological balance, treatment performance, and modeling approach, is discussed in Chapter Four. In addition, strategies to prevent clogging need to be evaluated for treating high strength wastewater to consider the utilization of a VSSF wetland. Clogging in the tropics is discussed in Chapter Five.

2.3.3. Closing the loop for water, energy, and food

Current agro-industrial and agricultural activities are based on linear economies. Under this model, water, energy, and food resources are unsustainably demanded and improper disposal of wastes provokes environmental degradation, which restricts resource availability. As an alternative, in a circular economy, the waste is considered as a resource. Anaerobic digestion can treat the biomass residues to obtain energy and fertilizers. The digestion process can be enhanced by promoting thermophilic processes. Solar energy can be capture through solar thermal collector to provide energy for heating digesters to thermophilic conditions. In Costa Rica, due to warmer conditions, solar thermal collectors can supply enough energy to maintain thermophilic digestion, which, when compared to mesophilic digestion, enhances solids degradation, yields more biogas, and releases pathogen-free digestate. The liquid digestate still has high concentration of solids, organic matter, and nutrients. To avoid water pollution, a HCTW can be implemented to reclaim water. Closing the loop by including an integrated system, such the one described, into current industries can decrease waste disposal into the environment and reduce demand for water, energy, and food.

2.4. Technical assessment of a sustainable development

Sustainability is a complex and broad concept, which varies in interpretation depending on the discipline and context. Multiple definitions can be found in literature. The 1987 Brundtland Report of the World Commission on Environment and Development defines sustainable development as *"development that meets the needs of the present without compromising the ability of future generations to meet their own needs"*. The United Nations defines sustainable development as *"the process that constrains resource consumption and* *waste generation to an acceptable level, makes a positive contribution to the satisfaction of human needs, and provides enduring economic value to the business enterprise"* (UN 2015). Both definitions imply that any action in present societies should not threaten the satisfaction of existing and future generations.

Multiple social and technical assessments are available to evaluate the sustainability of new systems or technologies. Social assessments are complex as they evaluate the interaction between consumers, society, and the environment. For social assessment of sustainability, collaboration of stakeholders is needed to determine the economic, social, and environmental benefits or affectations a new development would yield on the population. Social assessments are based on sustainable development indices, and interpretations largely vary depending on economic, social, and environmental evaluator's point of view.

In contrast, technical assessments of sustainability are based on quantifiable metrics with relatively objective interpretations. Metrics assess direct interactions between new developments or systems and society. Customer satisfaction, safety, security, efficiency, lifetime, and cost, are some metrics that can easily indicate the benefits or affectations a new development would yield on society. For example, exergy is a thermodynamic metric that describes system performance according to the second law of thermodynamics. Opposite to entropy, which measures the low-quality energy of materials in thermodynamic disequilibrium, exergy measures the quantity and quality of energy that a particular material possesses if it is brought into thermodynamic equilibrium (Jørgensen 2006, Rosen 2012, Querol, Gonzalez-Regueral et al. 2013). A system that converts materials with high entropy into high quality end products with low entropy will be sustainable if the balance between inputs (e.g.: wastes) and outputs (e.g.: biogas, fertilizers, and reclaimed water) is positive (Wall 2010, Woudstra 2016). Therefore, exergy serves as a powerful tool for improving sustainability as identification and implementation of systems that utilize wastes would reduce resource consumption and waste generation while providing beneficial outcomes from wastes to satisfy human needs. In thermodynamic terms, Hornbogen (2003) indicate that waste utilization reduces entropy in the environment at the time high quality end products (with high exergy) are yielded, increasing sustainability (Figure 2.6). This approach is used to evaluate sustainability is this dissertation.

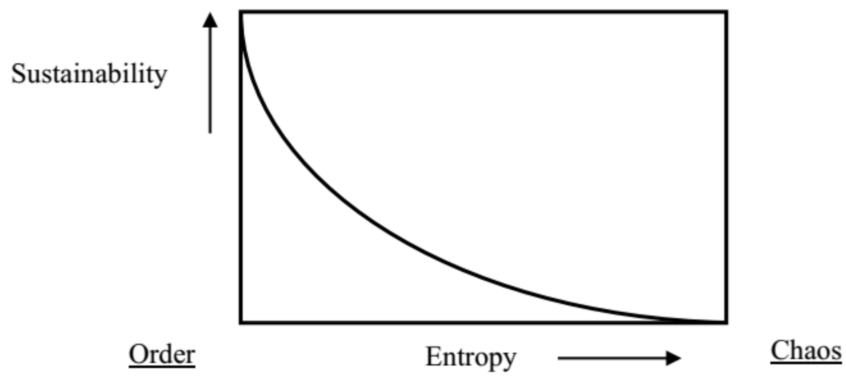


Figure 2. 6. Entropy, exergy and sustainability (Hornbogen 2003).

CHAPTER 3: TECHNICAL PERFORMANCE OF A SOLAR-POWERED WASTE UTILIZATION AND TREATMENT SYSTEM IN COSTA RICA

Abstract: Organic wastes are seen as residues, even though energy can be extracted from them. To simultaneously treat and utilize organic wastes, it is proposed to combine solar thermal collector, anaerobic digester, and constructed treatment wetland technologies. The goal of this study was to evaluate the technical performance of: 1) solar thermal collectors for maintaining thermophilic temperature in the anaerobic digester; 2) thermophilic anaerobic digestion for converting organic wastes into energy and fertilizers; and 3) VSSF-CTW for treating water from liquid digestate. In addition, a cash flow analysis was conducted to determine the payback period of the proposed system. Daily, the system utilizes 863 kg of mixed animal and food wastes to generate 263 MJ renewable energy, produce 28 kg nitrogen and phosphorus fertilizer, and reclaim 550 kg water. The net revenue, considering electricity and fertilizer production, is $2,146 annually. The payback period for the system is estimated to be 21 years. A sensitivity analysis demonstrated that by optimizing the total solids in the feed and reducing the solar thermal collector area, the payback period can be reduced to 9 years. The implemented system has successfully demonstrated a self-sufficient and flexible waste utilization and treatment system. It creates a win-win solution to satisfy the energy needs of the community and address environmental concerns of organic wastes disposal in the region.

3.1. Introduction

The agriculture sector, as the second largest industry in Central America, has contributed an average of 9.19% of the total Gross Domestic Product (GDP) of Costa Rica in the past decade

(EN 2015). Agricultural and agro-industrial activities generate a vast amount of organic wastes, such as animal manure, pineapple residues, sugarcane bagasse, rice straw, and coffee residues. Combustion of dry residues (i.e., sugarcane bagasse) and land application of wet wastes (i.e., animal manure), which are the most often used disposal approaches, have unfavorable economic performance and produce greenhouse gas (GHG) emissions, air pollution, and water pollution. For example, it is estimated that equipment costs for combustion ranges from 2,500 to 4,000 USD/kw; whereas, anaerobic digestion ranges from 1,650 to 1,850 USD/kw (IRENA 2012). On the other hand, if land application of wet waste is not properly managed, nutrient and carbon pollution becomes an environmental risk. Runoff and erosion can transport nutrients into water bodies and provoke eutrophication (Sutton and Joern 1992, Johnson and Eckert 1995). Then, volatilization of ammonia and emission of methane can occur within after land application (Sherlock, Sommer et al. 2002, Huijsmans, Hol et al. 2003, Safferman and Wallace 2015).

Instead, organic wastes, rich in proteins and high-caloric carbohydrates, are potential renewable resources for clean energy generation. Previous estimates suggest that approximately 600 MW electricity can be generated in Costa Rica from the agricultural residues each year (Coto 2013). However, only 18.9 MW of electricity is currently generated from organic residues, which is merely 0.73% of the total power capacity of 2,600 MW in Costa Rica (ICE 2016, Kohlmann 2016). Development and implementation of integrated technologies to treat agricultural wastes can create opportunities to alleviate negative environmental impacts of organic waste streams, increase access to affordable clean energy, and reduce GHG emission in rural communities in Costa Rica.

Anaerobic digestion is a natural biological conversion process that is proven effective at converting wet organic wastes into biogas and producing clean electricity while also reducing

GHG emissions and nutrients, carbon, and solid loads in the effluent digestate (AgStar 2010). Based on its operating temperature, anaerobic digestion can be categorized into thermophilic and mesophilic digestion. Thermophilic digestion occurs at a temperatures >45°C, while mesophilic digestion occurs at temperatures between 25 – 40°C. Numerous studies have demonstrated that thermophilic cultivation can enhance anaerobic digestion performance by shortening retention time, improving odor control, eliminating pathogens, increasing biogas production, and reducing total solids in waste streams (Aitken, Sobsey et al. 2005, Suryawanshi, Chaudhari et al. 2010, Sharma, Espinosa-Solares et al. 2013, Zarkadas, Sofikiti et al. 2015). However, thermophilic digestion requires input of thermal energy to maintain the temperature, which may lead to an unfavorable energy balance for small-scale operations (Vindis, Mursec et al. 2009). In contrast, large-scale operations can produce sufficient biogas to cover heating demand and maintain thermophilic temperatures while providing surplus energy for electricity generation (Zabranska, Dohanyos et al. 2002). However, the biogas produced is mainly used for heating purposes (Zabranska, Dohanyos et al. 2002). In order to overcome unfavorable energy balances for small-scale operations, other renewable energy sources, such as solar energy, need to be used (Suryawanshi, Chaudhari et al. 2010). Solar energy, an abundant renewable energy source in Costa Rica, is an excellent candidate to combine with small-scale thermophilic anaerobic digestion systems.

Several solar thermal conversion technologies have been developed, such as flat-plate solar thermal collectors, evacuated-tube solar thermal collectors, parabolic trough systems, power tower systems, and dish solar systems (Siva Reddy, Kaushik et al. 2013). Among these designs, flat-plate solar thermal collectors are simple and economical systems that are capable of efficiently providing the heat to maintain culture temperature during anaerobic digestion (EPA

1978, Alkhamis, El-Khazali et al. 2000). Furthermore, flat-plate solar thermal collectors are suitable for the tropics, since warm weather reduces heat loss and improves thermal efficiency. Integrating a simple solar collection method with anaerobic digestion technology may overcome unfavorable energy balances for small-scale thermophilic operations not only for rural Costa Rica, but also for other remote communities around Central America. In addition, the anaerobic digester can also play an important role of storing low-density and inconsistent solar energy (as heat) into a relatively dense and reliable biochemical energy source – methane (Tsalkatidou, Gratziou et al. 2009).

Even with the utilization and treatment of wastes provided by the solar-heated thermophilic anaerobic digester, the digestate still has relatively high levels of chemical oxygen demand (COD) (more than 10,000 mg/L) and nutrients (e.g., approximately 1,000 mg/L nitrogen and 200 mg/L phosphorus). Mechanical separation is widely adopted by anaerobic digester operations to separate the effluent into liquid and solid digestates (Monlau, Sambusiti et al. 2015). Solid digestate, which is rich in fiber and phosphorus, can be used as a fertilizer with enhanced nutrient retention in soils (Liedl, Bombardiere et al. 2006). As for liquid digestates, direct land application is common. However, further treatment to reclaim water from liquid digestate has attracted increasing attention (Carretier, Lesage et al. 2015, Sanyal, Liu et al. 2015), including treated liquid digestate prior to reutilization of treated water for irrigation, ground and surface water recharge, or process uses (e.g. washing floors of milking parlour).

Numerous studies have demonstrated that utilizing a constructed treatment wetland (CTW) to treat liquid digestate is an economically and technically sound approach to reclaim water (Denny 1997, Ritter and Shirmohammadi 2001, ITCR 2003, Kadlec and Wallace 2009). Free water surface (FWS) and subsurface flow are two typical CTW configurations. Compared to FWS-CTWs, subsurface flow-CTWs have the advantage of ensuring intensive contact between the wastewater and microbial biofilms growing on the media (ITCR 2003), thereby reducing the footprint of the wetland necessary to achieve treatment goals. Vertical subsurface flow-CTWs (VSSF-CTWs) are more common than FWS-CTWs for intermittent wastewater influents and, when surface fed, increase the aeration of the media (Kadlec and Wallace 2009). Therefore, a VSSF-CTW was incorporated into the integrated utilization system to treat the liquid digestate.

Costa Rica has many opportunities to increase the energy production from organic waste resources. One potential solution is the integration of anaerobic digestion and solar thermal technologies to simultaneously generate renewable energy and produce fertilizer. Besides the management of organic wastes, water reclamation is important. Thus, inclusion of a post-treatment technology is important for handling the liquid digestate, which otherwise could impair the environment if simple discharged into land or water bodies. Developed in Costa Rica, the goal of this study was to evaluate the technical performance of an integrated solar-powered anaerobic digester (SPAD) with a VSSF-CTW. In particular, this study evaluated the technical performance of: 1) the solar thermal collectors as the energy source for maintaining thermophilic temperature in the anaerobic digester; 2) the thermophilic anaerobic digestion as a technology for converting organic wastes into energy and fertilizers; and 3) the VSSF-CTW as a liquid digestate treatment technology for treating water. Finally, a cash flow compared the current system and a control system without thermal collectors, and was used to determine the payback period of the current system.

3.2. Material and methods

In 2011-2012, a solar-powered anaerobic digester and hybrid constructed treatment wetland (SPAD-HCTW) was installed at the University of Costa Rica (UCR) Fabio Baudrit Experiment Station (EEAFBM) located in Alajuela, Costa Rica (10.00 m N, -84.26 m W). In March 2013, the SPAD-HCTW started continuous operation. This study was conducted from August 2015 to March 2016, and constituted the first set of data continuously collected since operation began in March 2013.

3.2.1. System description

The SPAD included a modified flat-plate solar thermal collector, a thermophilic continuous stirred tank reactor (CSTR) anaerobic digester, and electrical generators (Figure 3.1). The modified flat-plate solar thermal collector converted solar energy into thermal energy to heat the influent of anaerobic digester and maintain the digester at thermophilic condition. A methane biogas storage bag served as the fuel storage. Solid effluent from the digester was composted and liquid digestate was post-treated by the HCTW, a VSSF-CTW working in series with a FWS-CTW. This study focused on the SPAD, thus only the VSSF-CTW has been included to demonstrate how these two technologies can create a close-loop system utilization of organic wastes and liquid wastes. Analysis of the HCTW and clogging status of the VSSF-CTW was not part of this study. Materials for construction of the SPAD-HCTW were bought at local suppliers in Costa Rica. The detailed individual units of this system are described as follow and shown in Figure 3.2.

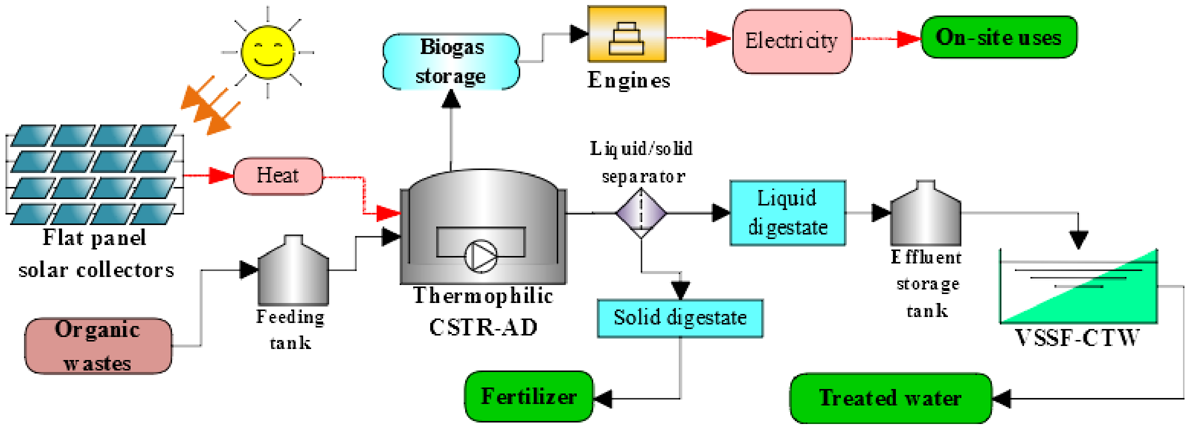


Figure 3. 1. Flowchart of the SPAD and VSSF-CTW. Mass flow is represented with a continuous line (-). Energy flow is represented with a hidden line (- -).

Figure 3. 2. Individual units of the system at the Fabio Baudrit Agricultural Station. a. Solar thermal collectors, b. Grinder, c. Feeding tank, d. Thermophilic CSTR-anaerobic digester (silver tank) and hot water tank (green tank), e. Liquid/solid separator, f. Effluent storage tank, g. Biogas bag, h. Engines, i. VSSF-CTW.

3.2.1.1. Solar thermal collection

The solar thermal collection unit aimed to provide sufficient thermal energy to maintain consistent thermophilic temperature in the anaerobic digester in the tropics. Thus, waste heat to maintain anaerobic digester temperature was not evaluated in this study. The solar thermal heating module consisted of a circulation pump (Model UP 26-99 F from Grundfos, Olathe, KS), a heat exchanger, and 36 m^2 of flat-plate solar thermal collector. Eighteen 2 m^2 thermal collectors (Termi-solar®, Costa Rica) were installed in three parallel rows of six collectors each row (Figure 3.2a). The average annual irradiance at the site was 10.2 MJ/m^2 (Wright 2008). The collectors were installed facing south at a 10° angle. Aluminum bronze (90/10) coils were used as the heat tubes in the solar thermal collectors. Water was the heat transfer fluid. The heated water was then stored in a 5 m^3 hot water tank (green tank in Figure 3.2d). A hot-water pump (Model PB 351MA from Wilo, Korea) circulated the hot water to heat the digester and maintain thermophilic temperatures (45 ± 2°C) using a 40 m High-Density-Polyethylene (HDPE) tubing heat exchanger in the digester.

3.2.1.2. Thermophilic CSTR anaerobic digester

The implemented system included an anaerobic digester tank, a feeding tank, and an effluent storage tank (Figure 3.2c, d, and f). All vessels were cylindrical tanks with flat bottoms made with HDPE. The effective volume of the digester was 20 m^3. The feeding and effluent tanks were 10 m^3 each. Food waste was transported from a nearby food distribution facility and consisted of non-commercial over-ripe or damaged vegetables and fruits, including cucumbers, peppers, avocado, papayas, pineapples, and tomatoes. Food wastes were ground by a grinder (Figure 3.2b) (model Leeson C 184K17FB150 from ICAFE ®, Costa Rica), mixed with chicken litter and treated water from the HCTW. This mixture was stored in the feeding tank. The chicken litter was collected from a chicken farm at EEAFBM. The food wastes (1.93 kg dry mass per day) and chicken litter (23.1 kg dry mass per day) were mixed at an average ratio of 1:12 (dry mass) with five cubic meters of water from the HCTW to target two percent of total solids (TS) in the feed. The characteristics of the feed are listed in Table 3.1. In the feeding tank, the mixture was mixed for 30 minutes per week by an external feeding tank pump (model AMT P/N 1626-305-00 from AMT, Royersford, PA). From Monday to Friday, one cubic meter with a total of 863 kg of the feed with an organic loading of 0.50 kg VS/m^3 digester volume/day was pumped (model AMT P/N 1626-305-00 from AMT, Royersford, PA) into the anaerobic digester from the feeding tank. The average hydraulic retention time (HRT) of the digestion was 20 days. The anaerobic digester was a thermophilic CSTR reactor with a submersible digester mixing pump (model 5763 from AMT, Royersford, PA) that mixed the anaerobic digester's contents for 10 minutes each hour. Biogas produced was collected in a biogas bag (HDPE 60 m^3 from Viogaz ®, Costa Rica). The biogas flow rate was measured using a biogas flowmeter (EKM-PGM 75 from EKM Metering, Santa Cruz, CA) installed on a pipeline connecting the digester to the biogas bag. A cubic meter of effluent from the anaerobic digester flowed by gravity when the anaerobic digester was fed. After digestion, a rotary liquid/solid separation unit (ICAFE ®, Costa Rica, particle size > 0.5 mm in diameter) was used to separate liquid and solid digestate from the anaerobic digester effluent (Figure 3.2e). The semi-solid digestate was used as fertilizer for on-site crop applications. The liquid digestate was stored in the effluent storage tank (Figure 3.2f). Finally, one cubic meter of the liquid effluent was discharged into the VSSF-CTW (Figure 3.2i) by gravity approximately daily from Monday to Friday.

Table 3. 1. Characteristics of the feed.

Parameters	**Mixture feed**
TS (g/L)	22.00 ± 3.30
VS (g/L)	11.60 ± 1.33
COD (g/L)	37.99 ± 2.75
Total carbon (% TS)	36.40 ± 1.30
Total nitrogen (% TS)	4.50 ± 0.20
Total phosphorus (% TS)	1.20 ± 0.11
pH	5.49 ± 0.12

Note. Data are the average of three replicates with standard deviation.

3.2.1.3. Control unit and data collection

A data acquisition system (DAQ model CR1000 Campbell Scientific, Logan, UT) collected data from thermocouples (type K, probe ungrounded) every 20 seconds for the feedback control to maintain the digestion temperature as ±2°C of the set temperature (i.e., 45ºC). The DAQ sent a digital signal to power the hot-water pump if the digester temperature was lower than the set temperature. The DAQ also recorded temperatures of the water at the flat-plate solar thermal collectors and the hot water tank every 5 minutes. In addition, the DAQ controlled a recirculation pump for the VSSF-CTW operation.

3.2.1.4. Electricity generator

The electricity generators were two 16 kW (Branco® B4T-5000 Bioflex, Brazil) biogas engines equipped with two activated carbon filters and a gas burner (Figure 3.2h). The biogas flowed from the biogas storage bag through the filters into the engines. The electricity generated from the engines was used to power pumps and other pieces of equipment in the system to satisfy operational requirements. Electricity usage for each piece of equipment was calculated based on duration of equipment operation (Table 3.2). Heat waste from electrical generators was not used for heating purposes as the thermal collection unit provided sufficient thermal energy to maintain consistent thermophilic temperature in the anaerobic digester.

Table 3. 2. Energy consumption per piece of equipment in the SPAD-HCTW.

Equipment	**Power**	**Time**	**Consumed energy**	**Schedule**
	(hp)	(h/week)	(kWh/week)	
Solar heating fluid transfer pump	0.17	49	6.0	7 hours/day, 7 days/week
Digester heating pump	0.46	43	15	6.27 hours/day, 7 days/week
Digester mixing pump	3.0	28	62	10 min/hour, 24/7
Feeding tank pump	2.0	0.5	0.75	30 min/week
Solid/liquid separator	1.0	0.85	0.63	10 min/day, 5 days/week
Grinder	5.0	0.50	1.8	30 min/week
Effluent pump	0.50	0.85	0.32	10 min/day, 5 days/week
Feed preparation pump	0.50	0.21	0.080	12 min/week
SPAD-HCTW exit pump	1.0	1.0	0.75	1 hour/week
VSSF-CTW recirculation pump	0.50	16.6	6.2	2.4 hours/day,7 days/week

3.2.1.5. VSSF-CTW

Liquid digestate usually is classified as a high strength wastewater and post-treatment is required to treat the water. Due to high concentrations of COD (more than 10,000 mg/L), TS (more than 5,000 mg/L), and TN (more than 1,000 mg/L) in the liquid digestate, this study evaluated a VSSF-CTW as the post-treatment unit. The VSSF-CTW (Figure 3.2i) was planted with *Cyperus papyrus*, *Iris graminea*, and *Canna indica* in 2012. Monthly, plants were trimmed to avoid expansion. By gravity, the VSSF-CTW received liquid digestate from the effluent storage tank. On the north side of the wetland, a PVC inlet pipe (diameter = 0.05 m) discharged liquid digestate into a 2 x 2 m geotextile membrane (GT 131 from Skaps, Athens, GA). The geotextile membrane has an apparent opening size of 0.30 mm, a flow rate per square meter of 0.102 m^3/s, and a permittivity of 2.20 s^{-1}. The cell is an inverse square pyramid and the dimensions of the VSSF-CTW are: 9 x 9 m of the bottom area, 12 x 12 m of the top area, and 1.1 m depth. The substrate media in the VSSF-CTW from bottom to top are: 0.2 m of stone (particle size of 12-20 mm), 0.2 m of pea gravel (particle size of 4-8 mm), and 0.7 m of coarse sand (particle size of 0.75-2 mm, 32% porosity). A recirculation pump (model WS V52 from Franklin

Electric, Fort Wayne, IN) was used to recirculate water from the bottom of the VSSF-CTW to the surface. The recirculation distribution system was 1.60 m above the sand. The recirculation distribution system consisted of a PVC network (diameter = 0.0127 m) of four upright fire sprinkler nozzles on each corner of a square (7.00 × 7.00 m) centered in the wetland that were obtained from a local hardware store (EPA, Alajuela, Costa Rica). The height of the recirculation spray was chosen to decrease interference of the redistribution spray by plants. The recirculation was carried out once per day from 2:00 am to 4:00 am to reduce losses of water due evapotranspiration.

3.2.2. Mass and energy balance

Mass and energy balance analyses were conducted based on the data from the operation during August 2015 to March 2016. The mass balance was used to describe the mass flow through the system including solid digestate, CH_4, CO_2, and water. The energy balance analysis was based on the mass balance and operational data. Energy input was defined as negative and energy output was defined as positive. Energy inputs corresponded to daily average electrical consumption from weekly operation, including pumps, grinder, and mechanical separator (Table 3.2). The SPAD with the VSSF-CTW was compared with a control thermophilic CSTR-anaerobic digester system without solar thermal collectors or post treatment. For the control system, a portion of the biogas produced would be required to maintain the thermophilic temperature of the digester. A hot water heating unit (replacing one of the engines in the studied system) is included to use biogas to maintain the digestion temperature. The heat input required for the system was calculated based on specific heat (equation 3.1), where Q is the energy input for heating the feed (MJ/d), c_p is the specific heat capacity of the feed (kJ/kg °C), ρ is the density of the feed (kg/L), and Δt is the change in temperature (°C). Energy was required to heat an amount of 1,000 kg of feed per day (one cubic meter per day, considering a feed density of 1,000 kg/m^3). The specific heat of the feed was estimated at 4.2 kJ/kg °C (Kosseva and Kent 2013). The average temperature in Costa Rica was approximately 20°C (IMN 2016). The operational temperature of the thermophilic CSTR was 45°C. Finally, since biogas was used for heat and electric energy generation, the heating value (55.5 MJ/kg CH_4) of the methane was used to calculate the energy output. The overall methane utilization efficiency was set at 90%.

$$Q = c_p \rho \Delta t \qquad [3.1]$$

3.2.3. Cash flow and payback period

A cash flow evaluation was conducted for the current system and compared with the control system. The capital expenditure (CapEx) and operational expenditure (OpEx) of the system operation were used for the cash flow evaluation. In addition, the evaluation included revenues produced by the operation of the system. The parameters used for the evaluation are described in Table 3.3.

Table 3. 3. Parameters considered for the cash flow evaluation.

System components	SPAD with the VSSF-CTW	Control system without solar thermal collector
Feedstock	1,000 kg per day with 2.2% TS	1,000 kg per day with 2.2% TS
Food wastes	No cost	No cost
Chicken litter	No cost	No cost
Solar panel	36 m^2 solar panel	None
Anaerobic digester	20 m^3	20 m^3
Digester technology	Thermophilic digestion	Thermophilic digestion
Loading rate (m^3/day)	1	1
TS of the feed (g/L)	22	22
Retention time (day)	20	20
Reaction temperature (°C)	46	46
Biogas utilization	Two 16 kw engines	One 16 kw engine for electric generation and a biogas burner for hot water heating unit
Land use	$113/$m^2$	$113/$m^2$
SPAD system (m^2)	180	106
VSSF-CTW (m^2)	144	144
Labor cost (operator)	20% of a full-time employee	20% of a full-time employee
Maintenance	Pumps, chemicals, and filters	Pumps, chemicals, and filters
Bioenergy, water, and fertilizer	--	--
Bioenergy	On-site electricity uses, compensating the energy demand	On-site electricity uses, compensating the energy demand
Fertilizer	On-site uses, compensating the fertilizer use	On-site uses, compensating the fertilizer uses
Water	Process uses	Process uses
Financial analysis	--	--
Inflation rate	3%	--
Depreciation	MACRS	--

The CapEx included the acquisition of assets that would have a useful life beyond the tax year, and that will bring benefits from the day-to-day operation of the system. In this study, the solar thermal collection unit (solar collectors, hot water pumps, pipes, and hot water tank), the feeding unit (feeding tank, grinder, and conveyor), the anaerobic digestion unit (vessel, pump, and gas meter), biogas utilization unit (gas bag and engines), digestate management unit (effluent storage tank and solid/liquid separator unit), treatment unit (VSSF-CTW, including media and pump), and system installation were considered as CapEx. The system installation was estimated as the 15% of the CapEx (MacDonald 2011). In addition, land utilized for installation of the system was considered in the CapEx. A total of 74 (solar panels area) of the 180 m^2 needed for the solar thermal collection unit were not considered for the control system. EEAFBM is located in a region where the land cost is $113 per m^2 (Hacienda 2016). The Modified Accelerated Cost Recovery System (MACRS) is a depreciation tool which allows for greater accelerated depreciation over longer time periods. The MACRS was used to calculate the annual depreciation of CapEx. The depreciation period was set at 20 years. The depreciation was estimated for the solar thermal collection unit, the feeding unit, the anaerobic digestion unit, biogas utilization unit, digestate management unit, and treatment unit. The system installation and land were not included in depreciation calculations. The annual depreciation rates from MARCRS were: 0.100, 0.188, 0.144, 0.115, 0.092, 0.074, 0.066, 0.066, 0.065, 0.065, 0.033, and 0.033 (after 10 years). Finally, feedstock (food wastes and chicken litter) was delivered on-site free of cost and was not included as costs. Thus, no cost of acquisition and transportation for the feedstock was considered. This case would be typical for farmers that have feedstock onsite.

OpEx included the expenses needed to run the system, such as labor and maintenance of equipment. The labor cost was based on the current wage rate in San Jose, Costa Rica (year 2015-2016) (UCR 2016). Operation of the SPAD consisted of preparing feedstock (one hour per week). Feeding the anaerobic digester, collecting solids from the solid/liquid separation, and other minor activities such as clean up took 1.5 hours per week. An additional two hours per week consisted of activities of overseeing discharge of liquid digestate effluent into the VSSF-CTW and harvesting plant biomass at the VSSF-CTW. In total, labor consisted of 0.9 hours per day, from Monday to Friday. Extra work time was needed due to unexpected maintenance of the system and, for safety, a total of 1.6 hours per day (20% of a full-time employee) was considered in this analysis for both current and control systems. Maintenance costs included repairs, replacement, and unexpected maintenance, and was estimated as the 3% annually of the CapEx

(MacDonald 2011). The system installation and land were not considered in the CapEx for calculation of maintenance costs. An annual inflation rate of 3% was set for OpEx based on the five-year (2010-2015) average inflation rate in Costa Rica (EN 2015).

Revenues included electricity offset and savings on fertilizer and water. Energy saving included the savings from both electricity and heat uses (such as heating the digester). The energy cost was $0.20/kwh equivalent according to the utility price in Costa Rica (year 2015) (ICE 2016). Fertilizer savings corresponded to the amount of fertilizer produced in the system. The fertilizer cost was calculated based on nitrogen and phosphorus contents in the solid digestate. The nitrogen and phosphorus costs based on the commercial fertilizer (year 2015) were used for the calculation. The cost of nitrogen fertilizer was $0.95/kg nitrogen and for phosphorus was $2.12/kg phosphorus (MEIC 2013). Water savings corresponded to treated water from the wetland that was used to replace the demand of fresh water for dilution, washing, and other uses for the SPAD. For the current operation, there was a minimum amount of the reclaimed water released from the system to the irrigation system. Thus, the cost savings for irrigation was not accounted for in the calculation. Similarly, an annual inflation of 3% was set for revenues.

The net cash flow based on depreciated CapEx, inflated OpEx, and revenues was calculated to determine the payback period. The payback period method indicates the length of time that the investment takes to payback costs based on the CapEx, OpEx, and revenues. In addition, a sensitivity analysis was conducted to elucidate effects of unit operations on the payback period of the system. Three key parameters, the solar collector area, TS of the feed, and wetland treatment area, were investigated. All the current values were adjusted by ±50% of their base values to elucidate their impact on changes of the payback period. The base payback period corresponded to the SPAD with the VSSF-CTW under the specified operational conditions. The biogas production was assumed to proportionally change with the TS change in the feed. The corresponding revenue from the energy savings was used as the base cost. The sensitivity analysis also assumed that wetland treatment would not be affected by changes in treatment area enough to prevent reuse of the treated water in the SPAD. Further analysis incorporating changes in treatment needs to be conducted to conclude more accurate results.

3.2.4. Analytic method

Weekly samples were collected from August 2015 to March 2016. Liquid samples were taken weekly from the feeding tank, the effluent storage tank, and the VSSF-CTW. Certified methodologies for collection of samples at the Water Quality Laboratory at the Research Center of Environmental Pollution, University of Costa Rica (CICA-UCR) were followed. Samples were collected using 1 L bottles, capped with a lid, and were kept at 4°C until analyses. Temperature, pH, COD, and TS (including VS and FS) were measured at EEAFBM. Temperature and pH were measured using a pH meter (model HI-2211 from Hanna Instruments, UK). Hach method #8000 and Hach method #8276 were followed for COD and TS, respectively. A DRB 200 reactor (Hach product #LTV082.53.40001) and a DR 900 multiparameter handheld colorimeter (Hach product #9385100) were used to digest and measure COD digestion vials (high range digestion vials from 0 to 1,500 mg COD/L, Hach kit). For TS, VS, and FS, samples were dried in disposable aluminum dishes (VWR®, catalog number 25433-008) for 24 hours in a StabilTherm gravity oven (model OV-12A from Blue M, East Troy, WI) at 100°C. Then, dishes were put in a desiccator and, after cooling for 30 minutes, were weighed in an analytical balance (Ohaus Corporation, Mexico). Then, samples were put in a StableTemp furnace (model CBFS516A from Cole-Parmer, Vernon Hills, IL) at 500°C for 30 minutes to determine FS and

VS. Additionally, total nitrogen (TN) and (total phosphorus) TP were analyzed at CICA-LCA laboratory. TN (method MAQA-40) and TP (method MAQA-1) methods followed the Standard Methods for the Examination of Water and Wastewater (Rice and Bridgewater 2012). Method 5310 B (modified) was followed for TN by a combustion method (TOC-V CSH/CSN from Shimadzu, Columbia, MD); method 4500-P D was followed for TP by a UV-visual spectrophotometer (Evolution 600 from Thermo Scientific, Madison, WI).

Feedstock (food waste and chicken litter), solid digestate (collected after the solid/liquid separator), and sediments (collected at the geotextile membrane) characteristics were measured at the Agronomy Research Center, at the University of Costa Rica (CIA). First, on-site, total feedstock, solid digestate, and sediments were weighed by an industrial scale (Romanas Oconi S.A., Costa Rica). Then, a sample of each one was taken to measure the TS (including VS and FS) of the feedstock, solid digestate, and sediments following Hach method #8276. Chemical composition (C and N) of the solids were analyzed at the CIA laboratory. Method SC09-LSF-P06 followed the Dunas method to determine N and C using an autoanalyzer (Vario Cube from Elementar, Philadelphia, PA).

Biogas samples were taken monthly using a sampling pump (SKC ® Grab Air, Bag Sampler Cat. No. 222-2301) and stored in gas sampling bags. Bags were kept at 4°C prior to the analysis at the Center for Research in Electrochemistry and Chemical Energy, University of Costa Rica (CELEQ). Biogas quality (CH_4 and CO_2 contents) was quantified using a gas chromatographic method (Hewlett Packard ® model HP6890 Plus, Littleton, CO) equipped with a thermal conductivity detector. The column was maintained at 250°C and argon was used as carrier gas. The injected sample volume was 100 µL and the syringe was purged three times before injection. All three laboratories, CICA, CIA, and CELEQ, follow standard methodologies accredited by the Costa Rican Accreditation Institute (ECA).

3.3. Results and discussion

3.3.1. Solar thermal collection

The temperature profile of the SPAD demonstrated that the heat transfer fluid (water) in the solar thermal collectors reaches a peak temperature of 82°C around noon (Figure 3.3). The solar thermal energy kept the temperature of the 5 m^3 hot water tank (green tank in Figure 3.2d) in the range of 50 to 78°C, which held enough thermal energy to maintain the anaerobic digester at a consistent thermophilic temperature of 46 ± 2°C (Figure 3.3). The temperature profile clearly indicated that 36 m^2 was sufficient for a\the solar collector to satisfy the thermal energy demand of a 20 m^3 thermophilic CSTR- anaerobic digester in Costa Rica. In contrast, an anaerobic digester assisted by solar power in a temperate region, still required biogas energy or waste heat from engines to maintain the digester temperature in winter months (Tsalkatidou, Gratziou et al. 2009). The stable year-round solar radiation and temperature in tropical areas were certainly beneficial to simplify design and implementation of solar-powered systems. Similarly, there is an advantage of utilization of solar thermal collection in the tropics if compared to an anaerobic digester heated by a solar greenhouse. In China, Hassanein, Qiu et al. (2015) evaluated a solar greenhouse, which basically consists on a greenhouse surrounding the anaerobic digester to keep warm temperatures, even during winter. The greenhouse maintained high enough temperature to perform biogas production in the anaerobic digester (28°C, 49°C, 33°C, and 16°C during spring, summer, autumn, and winter, respectively); however, those conditions were unstable for methanogens (Hassanein, Qiu et al. 2015).

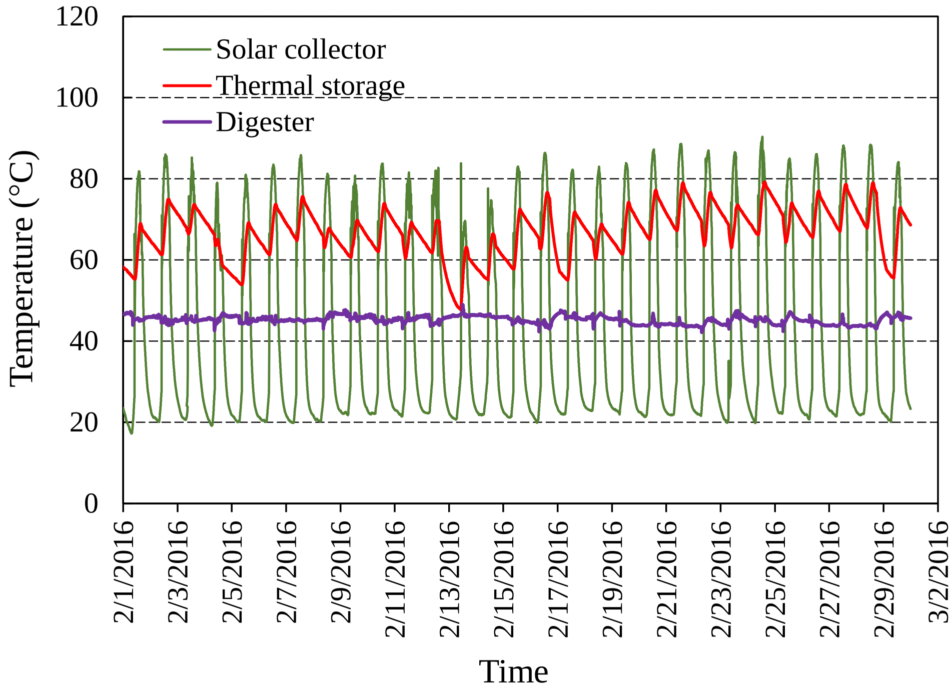


Figure 3. 3. Temperature profile of the solar thermal collection unit and thermophilic CSTR-anaerobic digester. Data for February 2016 are presented.

3.3.2. Thermophilic CSTR anaerobic digester

As the solar thermal collection unit maintained a temperature of 46 ± 2°C in the anaerobic digester, the CSTR- anaerobic digester was categorized as thermophilic. The thermophilic CSTR- anaerobic digester was fed, from Monday to Friday, with one cubic meter (or 863 kg) of feed with an organic loading of 0.50 kg VS/m^3 digester volume/day (HRT = 20 days and mixing time of 10 minutes per hour). Under this condition, the thermophilic CSTR-anaerobic digester had an average biogas production of 15.1 m^3 per day, with a corresponding CH_4 content of 68% (CH_4 production of 10.2 m^3 per day). The TS and VS were reduced to 0.99% and 0.47% in the anaerobic digester effluent from 2.20% and 1.16% in the feed, respectively. The TS and VS removal of 55% and 59%. Correspondingly, a CH_4 productivity of

1.42 m^3/kg VS reduced was achieved. After liquid and solid separation, the TS in the liquid digestate was further reduced to 0.47%. The solid digestate had 15.9% TS and contained 2.62 g TN/kg TS and 31.0 g TP/kg TS, which was used as a fertilizer for on-site applications. The pH of the digestion was steady at 7.89 and no external pH adjustment was needed, even though the feed had a relatively low pH of 5.45. The pH of the liquid digestate after the liquid/solid separation was at 7.95. The strong buffer capacity of the digestion indicated that stable and robust anaerobic microbial communities were established in the anaerobic digester. Zarkadas, Sofikiti et al. (2015) obtained similar VS reductions (53.3 – 73.9%) and pH values (7.8 – 8.2) during the co-digestion of food waste and cattle manure in a laboratory-scale thermophilic CSTR- anaerobic digester (reactor volume: 118 mL at 55°C). Biogas production was doubled by our CSTR- anaerobic digester, which produced 878 mL CH_4/g of VS added compared to 370 mL CH_4/g of VS added reported by the Zarkadas, Sofikiti et al. (2015). Feedstock characteristics and frequency of mixing can be impacting in the methane produced per gram of VS (Zarkadas, Sofikiti et al. 2015). In addition, a thermophilic (56°C) CSTR- anaerobic digester (volume capacity of 40 m^3) with a working volume of 27.4 m^3 treated 1.02 m^3/d of chicken litter-slurry (Bombardieri, Espinosa-Solares et al. 2007). Mixing was performed by biogas recycling (bubbling) and a pump that conveyed the digestate to the heat exchanger at a rate of five minutes each hour. The biogas production was 20.5 m^3/d; however the CH_4 content was lower (56.3%) compared to the value obtained in our study. Thus, CH_4 production was similar, 11.48 m^3/d for Bombardieri, Espinosa-Solares et al. (2007) compared to the 10.2 m^3/d in this study.

3.3.3. VSSF-CTW

Even though the SPAD converted a significant portion of VS into biogas, the liquid digestate from the digester still had very high nutrient contents. Liquid digestate concentrations of COD, TS, TN, and TP were 7,456, 5,200, 1,209, and 124.2 mg/L, respectively (Table 3.4). The color of the liquid digestate was still black (Figure 3.4). Additional post-treatment was needed to further reduce the nutrient load of the water and improve its irrigation quality. The VSSF-CTW was used for treatment as VSSF-CTW are effective for removing TS, COD, and NH_4 (Vymazal 2007, Kadlec and Wallace 2009, Abou-Elela and Hellal 2012, Bohórquez, Paredes et al. 2016). The VSSF-CTW was resilient to fluctuating loads and to low (rains lower than 1.91 cm/week) and high precipitation periods during this study. On average, from August 2015 to March 2016, the removal of COD, TS, TN, and TP by the VSSF-CTW was 96.7%, 73.1%, 90.8%, and 99.0%, respectively. Concentrations of COD, TS, TN, and TP in the treated water varied based on the precipitation period. During the high precipitation period, from August to December 2015, all concentrations except TS (543 mg/L) satisfied the discharge standard in Costa Rica (COD < 150 mg/L, TS< 50 mg/L, TN < 50 mg/L, and TP < 8 mg/L) (MINAE-MSP 2007) (Table 3.4). In contrast, during the low precipitation period, from January to March 2016, only TP (1.88 mg/L) met the discharge standard in Costa Rica (Table 3.4). First, the geotextile membrane removed 4.81% of COD, 27.9% of TS, 13.4% of TN, and 19.5% of TP. This membrane avoided direct contact of the liquid digestate with the filter media and allowed easy recovery of sediments. In fact, since March 2013, no crust formation has been observed in the filter media of the VSSF-CTW. During the high precipitation period, the treated water was used for primarily the SPAD and, to a lesser extent, for irrigation. For irrigation, the TS concentration did not represent a risk compared to usual TS concentration of manure when is applied as fertilizer. During the low precipitation period, the treated water was reused in the SPAD to replace the demand of fresh water for dilution, washing, and other uses. In addition, the VSSF-CTW unit demonstrated comparable performance with other studies, even with substantially higher concentrations of influent nutrients, which indicated that the CTW was an efficient process to treat the water from the liquid digestate. Barros, Ruiz et al. (2008) reported that using a CTW to treat the effluent from an up-flow anaerobic sludge blanket digester reduced COD, TN, and TP from 466, 55, 3.76 mg/L to 28, 37.5, and 2.8 mg/L, respectively (Barros, Ruiz et al. 2008). Comino, Riggio et al. (2013) used a VSSF-CTW to treat the diluted effluent of a CSTR digester. The concentrations of COD, TN, and TP in the treated water were 194.5, 9.42, and 0.26 mg/L, respectively, with corresponding removal of 76%, 91%, and 80% (Comino, Riggio et al. 2013).

Table 3. 4. Characteristics of the liquid digestate and treated water after the VSSF-CTW during both low and high precipitation periods.

Parameters	**Liquid digestate (n=25)**	**Treated water after the VSSF-CTW**	
		High precipitation (n=16)	**Low precipitation (n=9)**
COD (mg/L)	7,456 ± 571	66.50 ± 12.86	564.0 ± 120.2
TS (mg/L)	5,203 ± 440	543.4 ± 63.5	2,926 ± 272
VS (mg/L)	1,814 ± 261	171.2 ± 35.3	1,087 ± 154
TN (mg/L)	1,209 ± 82	34.04 ± 8.55	246.9 ± 28.2
TP (mg/L)	124.2 ± 16	0.8010 ± 0.1310	1.880 ± 0.75

Figure 3. 4. Change of water quality during the process.

3.3.4. Mass and energy balance

The mass balance for the SPAD with the VSSF-CTW is depicted in Figure 3.5. A cubic meter per day was fed to an anaerobic digester of 20 m^3. The HRT was 20 days. The anaerobic digester reduced 45% of TS and correspondingly produced 14.5 kg biogas containing 6.13 kg CH_4 and 8.43 kg CO_2 per day from 863 kg wet feed. Biogas was stored in a 60 m^3 gas bag for electricity generation. After liquid and solid separation, 28 kg per day of semi-solid digestate containing phosphorus and nitrogen were produced, which was used as a fertilizer for crop farming at EEAFBM. Liquid digestate (820 kg) was further treated by the VSSF-CTW. The VSSF-CTW reclaimed 550 kg of water per day which was used for irrigation or operation of the SPAD (Figure 3.5).

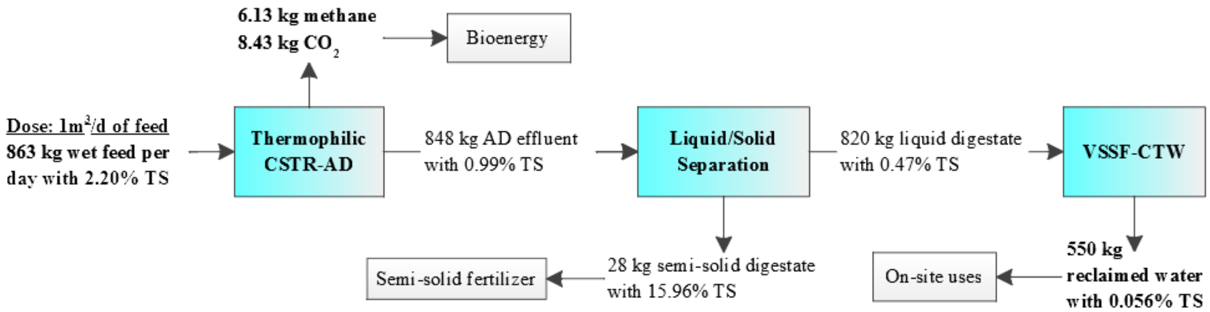


Figure 3. 5. Mass balance of the SPAD with the VSSF-CTW. Densities for CH_4 and CO_2 were estimated as 0.717 kg/m^3 and 1.84 kg/m^3.

The energy balance analysis for the SPAD with the VSSF-CTW and the control system without solar thermal collector is presented in Table 3.5. The energy balance analysis indicated that there was net energy output for the current system. The energy required to heat the feed and maintain the thermophilic temperature of the anaerobic digester was 126 MJ/day. The thermal solar collector unit provided sufficient energy to heat the feed and maintain the thermophilic temperature of the anaerobic digester, thus no external heat input was required for the SPAD. Instead, in the control system, a portion of the biogas produced would be utilized for the hot water heating unit to ultimately heat the feed and maintain thermophilic temperatures in the digester. The electricity input was electricity needed to power all equipment in the system (Table 3.2). In order to maintain a routine operation, 43 MJ/day of electricity was needed. The energy output from CH_4 combustion was 306 MJ/day. The electricity input for equipment and the energy output from methane were the same for both current and control systems. The net energy of the current system was 263 MJ/day, which was approximately twice as much energy produced by the control system without solar thermal collectors (127 MJ/day) (Table 3.5). The energy balance clearly demonstrated an advantage of use of solar thermal collectors to heat the SPAD.

Table 3. 5. Energy balance of the SPAD with the VSSF-CTW and the control system.

System	SPAD with the VSSF-CTW	Control system without solar thermal collector
Heat input (MJ/day)	0	-126
Electricity input (MJ/day)	-43.4	-43.4
Energy output (MJ/day)	306	306
Net energy (MJ/day)	**262.6**	**136.6**

According to the energy balance analysis of the system, it was apparent that the energy efficiency was enhanced by integrating solar thermal and anaerobic digestion technologies, as yielded biogas was not used for heating. Instead, Zabranska, Dohanyos et al. (2002) reported that, of a total 275 MWh/d produced, full-scale thermophilic CSTR- anaerobic digesters (12 anaerobic digesters, 4,800 m^3 each) required 60% of biogas produced for heating purposes and only 40% remained available for electricity production. Implementation of the SPAD and VSSF-CTW would also alleviate greenhouse gas emissions and ground/surface water contamination associated with current practices of organic residues handling in Costa Rica. For instance of the animal agriculture in Costa Rica, if all animal wastes (approximately 2,652,143 dry metric ton per year) from farm animals (Coto 2013) were treated by the studied system, 856,377 metric tons of CH_4 could be captured, 623,253 metric tons of dry solid digestate (as fertilizer) can be produced, and a net energy of 37 petajoule (349 MW electricity, approximately 18 times more electricity than current electricity produced by biomass in Costa Rica) can be generated each year.

3.3.5. Cash flow and payback period

A cash flow evaluation was carried out to examine CapEx, OpEx, and revenue of both current and control systems. As presented in Table 3.6, the CapEx of the current system implementation was $86,062. Among the CapEx components, land was the most expensive (42%

of the total CapEx). Then, among the operation units, the solar thermal collection (16%) was the most expensive, followed by anaerobic digestion (10%) and wetland treatment (8%). Thus, the CapEx for the control system was estimated to be 28% less costly than the current system from exclusion of the solar thermal collection unit and land (74 m^2 less). In addition, the system installation (15% of the CapEx) was less for the control system. Importantly, the heating system (a hot water heating tank replacing one of the engines) did not represent an additional cost for the current system.

The OpEx included both maintenance and labor costs, and the control system was estimated to be 12% less costly than the current system. The maintenance cost was estimated as the 3% annually of the CapEx. For maintenance, the CapEx did not include system installation and land costs. Thus, maintenance was estimated to be 31% less for the control system. This difference corresponded to exclusion of the flat-plate solar thermal collectors in the control system. In general, maintenance included oil changes of pumps, chemicals for biogas clean-up, filters for engines, sludge clean-up from the feeding tank and the effluent storage tank, PVC fittings (e.g., valves), and replacements of mechanical equipment. Life expectancy of mechanical equipment ranges from 7 to 10 years. Based on reported time operation in Table 3.2, the solar heating fluid pump, the digester heating pump, the digester mixing pump, and the VSSF-CTW recirculation pump should be prioritized for maintenance and replacement. Except the digester mixing pump, all pumps can be checked and maintained every week due to the easy access to them. In contrast, the digester mixing pump was installed inside the anaerobic digester and the only way to check it is by monitoring the recorded data at the DAQ. Yearly maintenance is required for this pump to avoid malfunctioning and alteration of anaerobic conditions in the anaerobic digester. All this maintenance required $1,290 per year (Table 3.6).

Labor costs corresponded to daily operation (from Monday to Friday) of the current system. The exclusion of the solar thermal collection unit and the inclusion of the heating system did not represent a difference in labor cost for both the current and control system. Daily routine focused on feeding and checking the system (1.5 hours per week), in addition to weekly preparation of the feed (one hour per day, only one day per week). Moreover, two hours per week consisted of activities such as overseeing discharge of liquid digestate effluent into the VSSF-CTW and harvesting plant biomass from the VSSF-CTW. These activities totaled 0.9 hours per day, from Monday to Friday. Since the system was relatively simple and automated, and considering a safety factor for unexpected maintenance, a worker spent 20% of his time every day (1.6 hours per day) for operation. Thus, the labor cost was $2,000 per year (based on wage rates in San Jose, Costa Rica, year 2015 (UCR 2016)) for both the current and control system.

Revenues considered energy offset from electricity produced from CH_4 combustion and savings on fertilizer. It was estimated that the current system generates $5,436 revenue per year, two times more than the control system. Incorporating solar thermal technologies, the biogas production can be doubled by promoting a thermophilic environment, without the need of external energy sources for heating (Suryawanshi, Chaudhari et al. 2010). In fact, the SPAD provided energy savings of $5,325 per year, more than 2 times the energy savings by the control system, which need a substantial portion of the biogas produced for heating the anaerobic digester mixture. Fertilizer savings were the same for both the current system and the control system. Fertilizer savings totaled to $111 per year, based on annual production of solid digestate (fertilizer) of 1,631 kg TS with 0.31% P and 0.026% N (mass balance, Figure 3.5). Fertilizers are produced regardless of the anaerobic digestion process. A net positive revenue of $2,146 per year was realized compared to the negative revenue (-$280 per year) from the control system.

Table 3. 6. Cash flow analysis.

System components	SPAD with the VSSF-CTW	Control system without solar thermal collector
CapEx		
Solar thermal collection unit	$13,500	-
Feeding unit	$4,500	$4,500
Anaerobic digestion unit	$9,000	$9,000
Biogas utilization unit	$4,500	$4,500
Digestate management unit	$4,500	$4,500
Treatment unit	$7,000	$7,000
System installation	$6,450	$4,425
Land use	$36,612	$28,250
Total CapEx cost	**$86,062 / 49,450***	**$62,175 / 33,925***
Revenue per year		
Energy saving	$5,325	$2,494
Fertilizer	$111	$111
Net Revenue per year	**$5,436**	**2,605**
OpEx per year		
Maintenance	$1,290	$885
Labor cost	$2,000	$2,000
Total OpEx cost	**3,290**	**2,885**
Net revenue		
Total net revenue	**$2,146 per year**	**($280)**

*: Total CapEx not including land costs.

Under the conditions presented in Table 3.6, the estimated payback period for the current system was 30 years. Land value ($113/m^2) in the region where EEAFBM is located substantially influenced the extended payback period obtained for the current system. Land costs for implementing the SPAD and the VSSF-CTW can be excluded in cases were the farm has sufficient room, such as dairy farms in rural areas. Excluding land costs, the total CapEx was $49,450 for the current system and the cash flow analysis indicated a payback period for the current system of 21 years (Figure 3.6). The payback period was reduced by 30% when the land cost was not considered.

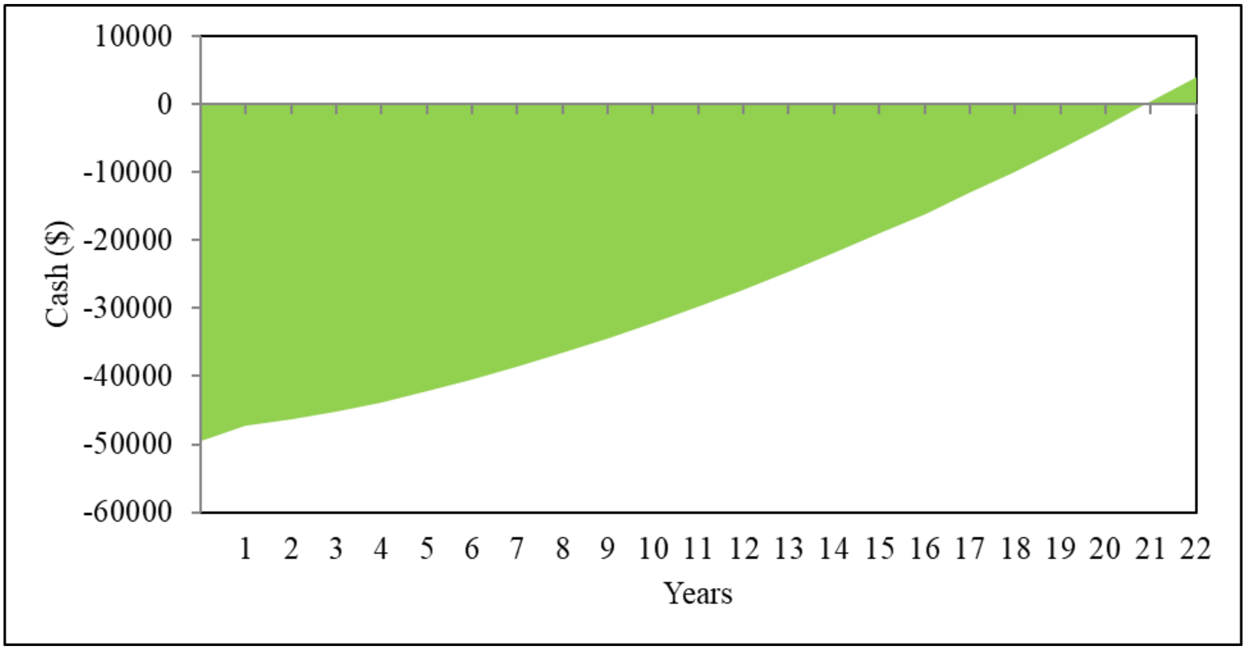


Figure 3. 6. The cash flow of the SPAD with the VSSF-CTW under the current operational conditions. Land costs were not considered.

Still, a payback period of 21 years is not attractive for commercial applicability. The system built was considered a full-scale experimental system and further studies on key parameters need to be conducted for scaling down the system based of client requirements. A sensitivity analysis was further carried out to delineate the impacts of three key parameters (solar collector area, TS of the feed, and wetland treatment area) on the payback period (Table 3.7). Results elucidated that TS of the feed was the most sensitive among the three parameters. A 50% increase of the TS of the feed could increase the CH_4 and fertilizer production to 15 m^3/day and 42 kg/day, respectively. The corresponding revenue was increased to $8,154/year. Under these conditions, the payback period can be reduced by 52% to 10 years. The second most sensitive parameter was the solar collector area. Removing 50% of the collector area can shorten the payback period by 12% to 18 years. The wetland area was the least sensitive parameter to influence the payback period. A 50% of area reduction only caused a 7% change of the payback period. The sensitivity analysis clearly indicated that the economic impact of the wetland was not as large as other unit operations on the stand-alone system. Consequently, decreasing the size of VSSF-CTW did not greatly affect the payback period; therefore, risking treatment performance of the VSSF-CTW by decreasing its size is not economically justified.

Table 3. 7. Sensitivity analysis of key unit operations on the payback period of the current system.

Unit operation	**Key parameter**	**Values**		**Corresponding base cost for the unit operation ($)**	**Change on payback period (%)**
		Base value	**Sensitivity range**		
Solar thermal collection	Solar collector area (m^2)	36	18-54	13,500	± 12
Anaerobic digestion unit	TS of the feed (%)	2	1-3	5,436	± 52
Treatment unit	Wetland treatment area (m^2)	100	50-150	7,000	± 7

According to the sensitivity analysis, several approaches can be adopted to improve overall net revenues. Increasing the TS content of the feed was certainly the best option to significantly increase the biogas production and reduce the payback period. Meanwhile, considering the fact that the current solar thermal energy exceeded the thermal need of the system (more collector area than needed) (Figure 3.3), reducing the collector area was the second option to improve the economic performance. Combined together, a 50% increase in TS and 50% decrease in solar collector area could lead to a relatively short payback period of 9 years, which would greatly improve the economic performance of the system.

3.4. Conclusions

Since March 2013, the system has accomplished the goal of producing energy, fertilizers, and reclaimed water from biomass residues; however, no data had been continuously recorded prior to this study. In order to evaluate the treatment performance of the SPAD-HCTW, this study analyzed data from August 2015 to March 2016 (28 weeks) to demonstrate how each separate unit of the system performed, as part of a closed-loop system to yield energy, fertilizers, and reclaimed water.

The solar thermal collector unit provided more than sufficient heat for maintaining thermophilic temperature in the anaerobic digester. Thus, external heating sources were not needed (e.g., biogas, heat waste from engines), even during nights. The solar thermal collectors simplified implementation of thermophilic anaerobic digestion in the tropics. The thermophilic CSTR- anaerobic digester performed in concordance to similar laboratory and full-scale thermophilic CSTR-anaerobic digesters. Importantly, VS reduction during the digestion resulted in high quality biogas (68% CH_4 content by volume) and CH_4 production (10.2 m^3/d). Solid digestate, recovered after digestion, was easily manageable for producing fertilizer. The energy balance demonstrated the benefits of integrating solar and anaerobic digestion technologies. Compared to a thermophilic CSTR- anaerobic digester heated with a hot water heating unit utilizing biogas, the SPAD doubled the energy production. Thus, the SPAD was self-sufficient in terms of energy demand by the system. The VSSF-CTW was resilient to fluctuating loads of high strength wastewater and, importantly, reclaimed water for reuse. The treatment performance was comparable to other CTWs that treated effluent from other types of reactors. The treated water was useful for irrigation and for replacing the demand of fresh water for dilution of the feed.

From the technical point of view, the SPAD-HCTW demonstrated one approach for treating biomass residues and wastewater in Costa Rica. The sensitivity analysis demonstrated that optimization of parameters can reduce the payback period. Therefore, more studies to optimize thermal solar collector area, feedstock characteristics, feeding frequency, mixing frequency and others would be important, particularly since this study considered a limited time (August 2015 to March 2016, 28 weeks) outside of which other external factors could impact in the technical performance. A country, such as Costa Rica, that sells itself as a green country for ecotourism, needs to be aware of the treatment of organic wastes, and the SPAD-HCTW can turn an environmental and economic liability into a public and private asset.

3.5. Acknowledgments

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CHAPTER 4: PERFORMANCE OF THE HYBRID CONSTRUCTED TREATMENT WETLAND TREATING DIGESTATE EFFLUENT IN COSTA RICA

Abstract: Integrated into the landscape, constructed treatment wetlands are impacted by climatological conditions. Constructed treatment wetlands are attractive options for wastewater treatment because they have low costs of construction, maintenance, and operation; additionally, in tropical regions, wetlands are expected to have relatively consistent treatment year-round due to consistently warm temperatures. However, there are limited studies monitoring performance of constructed treatment wetlands in the tropics. Thus, there is a need to evaluate the treatment performance of wetlands in the tropics, particularly with respect to rainy and dry seasons. This study evaluates a hybrid constructed treatment wetland (HCTW) in Alajuela, Costa Rica that treats high-strength liquid digestate from a thermophilic anaerobic digester. The specific goals were to 1) evaluate treatment performance of the HCTW during rainy and dry periods and 2) evaluate the applicability of loading charts and contaminant removal models commonly used to describe temperate wetlands to model treatment performance of the tropical HCTW. This study was conducted during a low precipitation period (August to December 2015) and a high precipitation period (January to March 2016). During these two periods, the HCTW demonstrated sufficient storage capacity, even when unexpected runoff entered the system during the high precipitation period. Slightly better treatment of chemical oxygen demand (COD) and total solids (TS) occurred during the high precipitation period, likely due to dilution from precipitation. However, removal of total nitrogen (TN) and total phosphorus (TP) were similar for both the low and high precipitation periods. Effluent from the HCTW was either reused to dilute feed for the anaerobic digester (during both periods) or used for irrigation

(during the rainy period). The performance of the HCTW was good; however, more data would be needed to narrow central tendencies represented by loading charts and contaminant removal models.

4.1. Introduction

Worldwide, treatment of wastewater is often neglected in developing countries. It is estimated that over 80% of wastewater worldwide goes untreated, so that more than two million cubic meters of wastewater are discharged into the world's rivers, lakes, and oceans every day (Otchet 2003, UNESCO 2015). Costa Rica, located in Central America, is not an exception to this global problem. Limited financial commitments and a lack of enforcement of regulations endanger fresh water resources in Costa Rica (GWP 2012, Echeverria and Cantillo 2013). Only 5.81% of the 0.0860 km^3/year of domestic wastewater is treated in Costa Rica (Sato, Qadir et al. 2013). Additionally, industrial wastewater is rarely treated; for example, only five percent of 3,500 industries at the *Rio Virilla* watershed (0.3% of the Costa Rican territory) treat industrial wastewater (UN 2009, Coto 2013, Echeverria and Cantillo 2013). Therefore, there is an urgent need for innovative approaches for wastewater treatment, not only in Costa Rica, but in undeveloped and developing countries worldwide.

Constructed treatment wetlands (CTWs) can be used to treat multiple types of wastewater. Since 1950, CTWs have been used to treat domestic, animal, mining, and industrial wastewaters, liquid leachates, urban stormwater, and field runoff (Kadlec and Wallace 2009). Based on the water flow regime, different engineered designs, such as vertical subsurface flow (VSSF), horizontal subsurface flow (HSSF), free water surface (FWS), and hybrid constructed treatment wetlands (HCTWs), aim to stabilize, sequester, accumulate, degrade, metabolize, and mineralize nutrients in the wastewater, emulating natural wetlands (Halverson 2004, Vymazal 2007). CTWs are resilient to fluctuating loads, require low technical expertise to operate, typically require less time for operation and maintenance, and are typically lower in costs than conventional wastewater treatment (Kadlec and Wallace 2009, Konnerup, Koottatep et al. 2009). The longevity of CTWs varies depending on the system; however, estimations vary from 10 to 15 years (Wallace and Knight 2006). There are limitations to use of CTWs for decentralized wastewater treatment. Subsurface-flow CTWs require preliminary treatment of the wastewater to remove sediments that would clog the wetland; additionally, CTWs typically require larger areas than centralized wastewater treatment facilities (Dallas, Scheffe et al. 2004, Langergraber 2007).

The hydrological balance is an important component of CTW design, including estimation of the saturated water depth in the wetland. The saturated zone plays an important role in wastewater treatment as it conveys the pollutants in the wastewater through the filter media or vegetation, including the root zone (Kincanon and McAnally 2004). Therefore, a hydrological balance of the wastewater inflows, outflows, and storage within the CTW is crucial to determining the capacity of a CTW, the hydraulic loading rate (HLR), and the hydraulic retention time (HRT). Equation 4.1 shows the inflows and outflows for the hydrological balance of a CTW. Q_i (m^3/d) is the influent flow, Q_o (m^3/d) is the effluent flow, Q_c (m^3/d) is the influent flow due to runoff, Q_f (m^3/d) is the loss of flow due to filtration, and Q_{gw1} and Q_{gw2} (m^3/d) are the flow additions and losses due to groundwater discharge or recharge. P (m/d) is the precipitation depth, which is multiplied by the CTW surface area, A (m^2). $ET(A)$ (m^3/d) is the evapotranspiration losses (in height) multiplied by the CTW surface area. D_s (m^3/d) is the change in volume of water stored in the CTW (Kadlec and Wallace 2009).

$$Q_i - Q_0 + Q_c - Q_f - Q_{gw1} + Q_{gw2} + P(A) - ET(A) = D_s \qquad [4.1]$$

Usually, CTWs are lined, either by installing a geomembrane or using a compacted clay layer. Therefore, Q_f, Q_{gw1} and Q_{gw2}, are zero, and equation 4.1 takes a simplified form (equation 4.2).

$$Q_i - Q_0 + Q_c + P(A) - ET(A) = D_s \qquad [4.2]$$

Contaminant removal models (e.g.: black-box models) are commonly used for the design of CTWs (Reed, Ronald et al. 1995, EPA 2000, Shepherd, Tchobanoglous et al. 2001, Kadlec and Wallace 2009). These models do not aim to accurately delineate treatment processes, but serve as a simple tool for designing CTWs based on influent concentrations, HLR (or HRT), and pseudo-first-order areal rate coefficients (k). Pseudo-first-order areal rate coefficients depend on temperature, pollutant concentration, and other factors (Kadlec and Wallace 2009). These coefficients indicate an empirical rate of pollutant removal due to the sum of all biological and physicochemical processes that occur in the CTW (Stein, Biederman et al. 2006). The Arrhenius relationship describes the rate coefficient's dependency on temperature (equation 4.3), where k_T is the rate constant at a specified temperature T (d^{-1}), k_{20} is the rate constant at 20°C (d^{-1}), θ is a dimensionless temperature coefficient, and T is the temperature (°C). The dimensionless temperature coefficient indicates the influence of the water temperature on the removal of pollutants. If $\theta = 1.0$, the water temperature does not influence in the pollutant removal. But, if $\theta > 1.0$ the pollutant removal rate k_T increases with increasing water temperature. In contrast, k_T decreases with increasing water temperature if $\theta < 1.0$ (Kadlec and Knight 1996). Reported values for θ for constructed treatment wetlands range from 0.946 to 0.985 for biological oxygen demand (BOD), 1.001 to 1.062 for TN, 1.005 to 1.072 for ammonium (NH_4), 1.043 to 1.177 for nitrate (NO_3).

$$k_T = k_{20} \times \theta^{T-20°C} \qquad [4.3]$$

For design purposes, k values serve as an approximation. Kincanon and McAnally (2004) suggest to design using conservative (low) k values, as the microbial activity in the proposed CTW is still unknown. Usually, the k value used for design corresponds to the coldest month. Four contaminant removal models that rely on k or similar pseudo-first-order rate coefficients are the plug-flow, the k-C^*, tank in series (TIS (P k–C^*)), and time-dependent retardation (TDR) models (Table 4.1).

Table 4. 1. Contaminant removal models for designing CTWs.

Model approach	**Governing equation**	**References**
Plug-flow	$\frac{C}{C_i} = e^{-K_T \times t}$ [4.4]	(Reed, Ronald et al. 1995)
Modified k–C^*	$\frac{C-C^*}{C_i-C^*} = e^{-K_T \times t}$ [4.5]	(Kadlec and Wallace 2009)
TIS (P k–C^*)	$\frac{C-C^*}{C_i-C^*} = \frac{1}{\left(1+{k}/{Pq}\right)^P}$ [4.6]	(Kadlec and Wallace 2009)
TDR	$\frac{C}{C_i} = exp\left[\left(\frac{-k_0}{b}\right) ln(b\tau + 1)\right]$ [4.7]	(Shepherd, Tchobanoglous et al. 2001)

Notes. C is effluent concentration (mg/L), C_i is influent concentration (mg/L), C^* is background concentration (mg/L), K_T is the rate coefficient at a specified temperature T (d^{-1}), t is the hydraulic retention time (d), k is the modified first-order areal constant (m/d), P is the apparent number of tanks, q is the hydraulic loading rate (m/d), k_0 is the initial first order volumetric rate constant (d^{-1}), b is the time based retardation coefficient (d^{-1}), and τ is the retention time (d).

In the plug-flow model (equation 4.4, Table 4.1), the CTW is considered an attached growth biological reactor, in which removal is irreversible, first-order, and homogeneous. This model predicts the pseudo-first-order removal of individual pollutants (e.g., BOD, COD, and TS) that are not part of multi-step decomposition processes (e.g., nitrogenous compounds) (Kincanon and McAnally 2004). The plug-flow model assumes that all of each pollutant can be completely degraded in the CTW. In contrast, the modified k–C^* model (equation 4.5, Table 4.1) includes the C^* parameter, which is the residual or background concentration of the pollutant in the CTW. The background concentration, C^*, accounts for nondegradable compounds or minimum values of nutrients due to plant senescence at low nutrient concentrations – both phenomena which result in a non-zero lower limit that is approached asymptotically (Kadlec and Wallace 2009). Similarly, the tank in series (TIS) model (equation 4.6, Table 4.1) uses parameters k and C^*, but also includes the parameter P, which is the apparent number of TIS. Since the assumption of a homogeneous attached growth biological reactor with ideal plug flow is not realistic within a wetland (Kadlec 2000), the TIS approach accounts for dispersion within the wetland. As the number of tanks approaches infinity, the CTW acts as a plug flow reactor, while a wetland with P=1 behaves as a continuous stirred tank reactor (CSTR). The C^*, for COD in subsurface CTW, can be estimated by using an empirical relationship, equation 4.8 (Kadlec and Wallace 2009); however, Stein, Biederman et al. (2006) indicated that C^*changes seasonally as microbial decomposition changes with the temperature.

$$C^* = 3.5 + 0.053C_i \qquad [4.8]$$

This unsteady behavior of C^* introduces more noise into the contaminant removal modeling. The time-dependent retardation (TDR) model (equation 4.7, Table 4.1), by Shepherd, Tchobanoglous et al. (2001), assumes that the removal rates of organic matter decrease with the percentage of biodegradability of the wastewater, from relatively fast rates for easily biodegradable constituents to relatively slow rates for less biodegradable constituents; time dependence of removal rates is used in place of a constant C^* value. Therefore, the TDR model considers changes in the removal rate by replacing the C^* in the k–C^* model with two parameters, K_0 and b. Shepherd, Tchobanoglous et al. (2001) obtained k_0 values from 9 to 12 d^{-1} and b values from 2 to 5 d^{-1}, which allows a steady decrease in COD rather than an asymptotical approach to a unique C^*value. However, this model was developed for one pilot-scale CTW planted with bulrush and cattails; K_0 and b have not been determined or the model reproduced for full scale systems.

Loading charts, or graphical representations of output concentrations versus input loading rates, developed from a large number of empirical data sets are an alternative to contaminant removal models for design purposes. Loading charts based on organic loading rates (OLR, $g/m^2/d$) and influent concentrations (mg/L) are used to estimate the surface area required for desired treatment. Empirical data has also been used to prepare graphs of output versus input concentrations. The slope of output versus input concentrations indicates the potential treatment performance of a certain type of CTW to reduce a particular contaminant (Kadlec and Wallace 2009). However, estimating surface area with this second approach is not possible due to the lack of area and water flow information (Kadlec 2009, Kadlec and Wallace 2009). These two graphical approaches show a central tendency of a treatment performance for a particular type of CTW; however, graphs should be used with caution because of considerable variability within and between systems. First, treatment performance within an individual CTW is variable due to seasonal and year-to-year changes (plant communities, internal hydraulics, and weather conditions). Second, there is intersystem variability in the treatment performance due to plant type, cell configuration (e.g., system geometry, depth, and substrate), and climate (Kadlec 2009, Kadlec and Wallace 2009).

Models and loading graphs have been developed to describe wetland treatment performance for most standard wastewater parameters. Contaminant removal models have been developed for TSS, BOD/COD, total phosphorus (TP), total nitrogen (TN), NH_4, NO_3, and fecal coliform (Table 4.1). Reputable loading graphs are available for HSSF-CTW of organic N, ammonia, TN, TP, fecal coliforms, TSS, BOD, and Total Kjeldahl Nitrogen (TKN), while loading graphs for VSSF-CTW are only available for TSS, BOD, and TKN (Kadlec and Wallace 2009). However, all this information was collected from temperate climates and the use of these data to design CTWs in the tropics might contribute to incorrectly sized CTWs.

Principle component analysis (PCA) is a multivariate statistical tool that has been used to elucidate if one or more parameters affect the treatment performance of wetlands. For example, Lee and Cha (2015) conducted a PCA to identify key meteorological parameters that might impact treatment efficiency of a CTW for stormwater treatment. For one year, water samples were collected after storm events at the influent and effluent of a FWS-CTW receiving runoff from agricultural lands in Naju city, Korea. Water quality parameters analyzed were TSS, BOD, COD, TN, and TP. Results indicated that the first three principal components described 73.5% of the total variance, in which rainfall depth, rainfall intensity, and antecedent dry days were identified as key meteorological parameters. In addition, Dong and Reddy (2010) conducted PCA to identify that, as TKN, phosphate (PO_4), and COD concentrations in the wastewater decreased across the wetland, the diversity and richness of microorganisms also decreased. These water quality parameters were found in the first two principal components, which explained the 51% of the total variance. Moreover, PCA can be used as a tool to identify which parameters are more mathematically robust to explain most of the variability in the data set, including independent and dependent parameters. This second approach simplifies identification of parameters that explain what is occurring in the treatment performance of the wetland.

CTW in temperate climates do not perform at the same efficiencies year-round due to winter dormancy of macrophytes and microorganisms (Healy, Rodgers et al. 2007). Temperature drastically changes due to seasonal variations in temperate regions, and temperature has been identified by several authors as one of the important parameters determining treatment performance (Bachand and Horne 2000, Kadlec and Reddy 2001, Stein, Biederman et al. 2006). Microbial communities typically have maximum growth rates in spring and later in midsummer. Microbial activity is driven by temperature and impacts wetland's treatment performance. For example, Bachand and Horne (2000) reported a reduction of nitrification rate from 1.1 to 0.10 g N/m^2d due to a change in the water temperature from 26 to 12°C; however, this is inconsistent with reported θ values. This discrepancy may be due to changes in environmental conditions other than temperature, such as shorter light periods and lower humidity in winters; additionally, temperature may also indirectly affect treatment, as it also influences solubility of oxygen, plant activity, and flow of partially frozen wastewater.

As the performance of CTWs is impacted by climatological conditions, modification of temperate-climate design approaches for CTWs in the tropics may be needed to maximize performance while minimizing costs. Studies indicate greater removal of pollutants in tropical CTWs than in temperate wetlands, most likely due to steady warm temperatures year-round in the tropics (Trang, Konnerup et al. 2010, Dan, Quang et al. 2011, Kelvin and Tole 2011, Zhang, Jinadasa et al. 2015). However, wetland's treatment performance might be impacted by prolonged and intense rainfalls in the tropics. Avila, García et al. (2016) indicated that rains could impact treatment efficiencies. Initial rains of the rainy season could flush pollutants from the surrounding area into wetlands; however, other studies have found that water inputs by runoff and precipitation dilute effluent concentrations (Suárez and Puertas 2005, Avila, Salas et al. 2013). This study aims to assess how the performance of a HCTW, which consisted of a VSSF-CTW and FWS-CTW in-series, responded to periods of high and low precipitation. Additionally, loading charts and contaminant removal models were evaluated for their applicability to describe the treatment performance of the HCTW.

4.2. Material and Methods

In 2011-2012, a solar-powered anaerobic digester and hybrid constructed treatment wetland was designed and built at EEAFBM, in Alajuela, Costa Rica (10.00 m N, -84.26 m W). In March 2013, the system began continuous operation. This study was conducted with data collected from August 2015 to March 2016, and focused on the hybrid constructed treatment wetland receiving wastewater from the anaerobic digester. Details and specifications of the solar-powered anaerobic digester can be found at Aguilar Alvarez, Bustamante Roman et al. (2016).

4.2.1. System description

The CTW was fed with liquid digestate from the solar-powered anaerobic digester system (SPAD) treating chicken litter and food waste. Effluent from the digester was separated mechanically into liquid and solid streams with a rotary screen separator (Solid/Liquid separator) (ICAFE ®, particle size > 0.5 mm in diameter). The solid digestate was used as a fertilizer for on-site farming uses. The liquid digestate was stored in an effluent tank and, at a rate of one cubic meter per from Monday to Friday, was discharged by gravity into the HCTW to be further treated. The HCTW consists of a VSSF-CTW followed by a FWS-CTW operating in-series. Treated water was used for irrigation or reused in the anaerobic digester (Figure 4.1). The design of the HCTW considered Healy, Rodgers et al. (2007) and Kadlec and Wallace (2009) approaches for intermittent sand filters and wetlands for wastewater; however, space constraints and research goals ultimately determined the size of the HCTW.

Figure 4. 1. Aerial image of the HCTW. Satellite image was taken from Google Earth.

The VSSF-CTW dimensions are 12.0 × 12.0 m at the top and 9.00 × 9.00 m at the bottom, with a median treatment area of 138 m^2 (Figure 4.2). The height of the wetland is 1.10 m and the slope of the walls is 27° with respect to the horizontal. The bottom and walls of the VSSF-CTW are lined with clay. The media is predominantly coarse sand (particle size of 0.75 – 2.0 mm) with a porosity of 32%. Below 0.70 m of the sand is 0.20 m of pea gravel (particle size of 4 – 8 mm) and 0.20 m of stone (particle size of 12 – 20 mm) to facilitate collection of the treated wastewater. The VSSF-CTW has a maximum water storage capacity in the media of 30.9 m^3, with an additional 80 m^3 of storage above the media. A PVC inlet pipe (diameter = 0.05 m) is located at the north side of the cell. One cubic meter per day of liquid digestate from the effluent storage tank flows through the inlet pipe into a square 4-m^2, 0.25-m deep trench of that is covered with a geotextile membrane (GT 131, Skaps, Athens, GA). The geotextile membrane has an apparent opening size of 0.30 mm, a flow rate per square meter of 0.102 m^3/s, and a permittivity of 2.20 s^{-1}. In the bottom stone layer, a distributed grid of drainage pipe (diameter = 0.075 m) collects and conveys the treated effluent to the VSSF-CTW well pump. In the well, a submersible pump (model WS V52 from Franklin Electric, Fort Wayne, IN) recirculates the treated effluent to the same cell at a flow rate of 110 L/min. Each weekday, recirculation occurred from 2:00 am to 4:00 am to avoid high losses of water from evapotranspiration. Recirculated effluent is sprayed from four sprinklers located 1.60 m above the ground and at each corner of a square (7.00 × 7.00 m) centered at the center of the wetland. From Monday to Friday, 1-m^3/d of the treated effluent from the VSSF-CTW was transferred to the FWS-CTW. The VSSF-CTW was planted with *Juncus effuses* (common rush), *Coix lacryma jobi* (Job's tears), *Cyperus papyrus* (papyrus), *Iris graminea* (dwarf iris or grass-leaved iris), and *Canna indica* (wild canna lily or African arrowroot) in 2012. After three years, *Juncus effuses* and *Coix lacryma jobi* had not survived, and at the time of this study, only *Cyperus papyrus*, *Iris graminea*, and *Canna indica* were thriving in the wetland. In August 2015, it was observed that *Cyperus papyrus* dominated the wetland (Figure 4.2). To avoid expansion of *Cyperus papyrus* across the wetland area and to maintain equal areas of each species, plants were trimmed monthly.

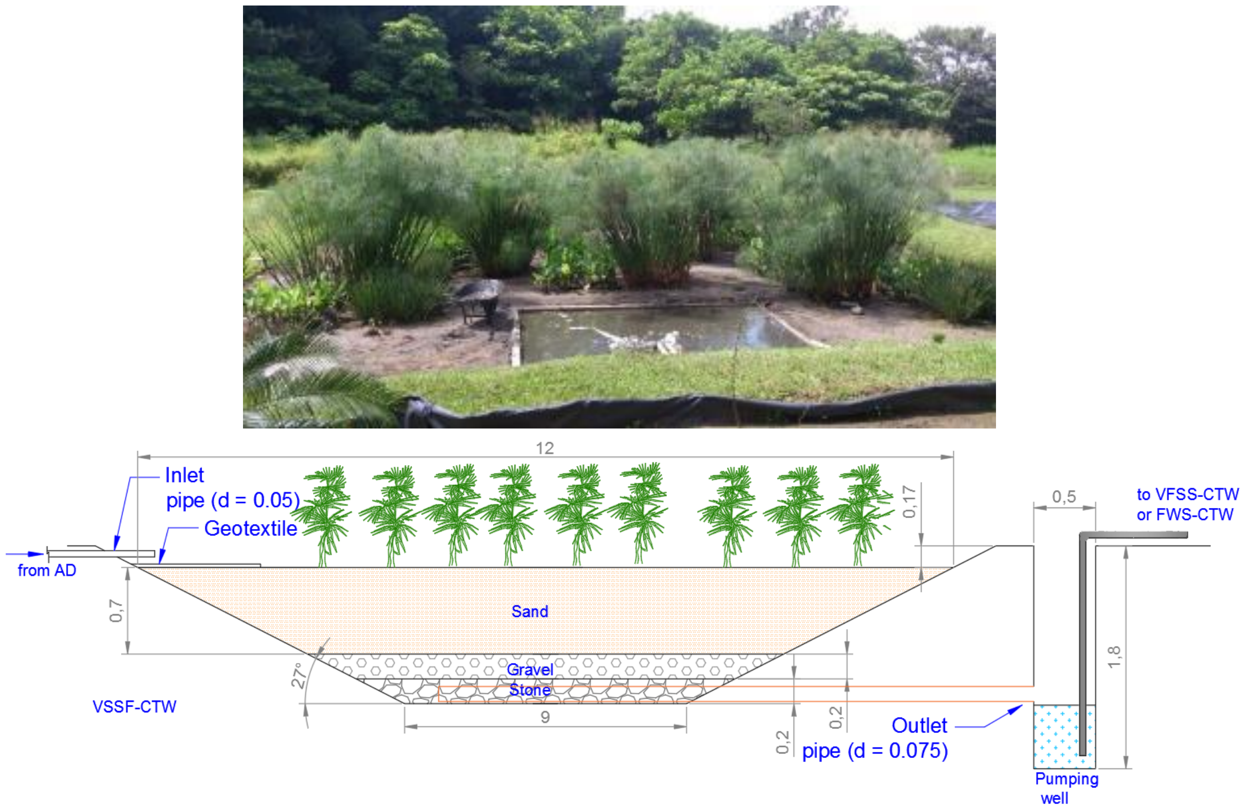

Figure 4. 2. Photo and schematic of VSSF-CTW. Dimensions are in meters. Picture was taken in August 2015.

The FWS-CTW (Figure 4.3) has the same dimensions as the VSSF-CTW. A layer of clay was used to line the bottom and walls of the cell. The maximum storage capacity is 152 m^3, which is the total volume in the cell (138 m^2 × 1.10 m deep). From Monday to Friday, the FWS-CTW receives one cubic meter of treated effluent from the VSSF-CTW. A PVC inlet pipe (diameter = 0.05 m) is located at the north side of the cell. The effluent pipe is located 2.73 m from the edge of the HSSF-CTW at a depth of 1.09 m above the bottom. The effluent pipe, a PVC pipe (diameter = 0.075 m), conveys the treated water to the FWS-CTW well pump. Similar to the VSSF-CTW, a submersible pump (model WS V52 from Franklin Electric, Fort Wayne, IN) recirculates the water from 2:00 am to 4:00 am every weekday. The recirculated effluent is discharged from a PVC pipe (diameter = 0.012 m) located in the East side of the wetland. The exit pump (model WS 102 from Franklin Electric, Fort Wayne, IN) pumps FWS-CTW effluent to either the digester feed tank or the station's irrigation system. The FWS-CTW is planted with two species of floating plants, *Eichhornia crassipes* (water hyacinth) and *Pistia stratiotes* (water lettuce) (Figure 4.3). Floating plants were divided by species by a frame of bamboo (3 m wide and 9 m length per species), and covered almost 100% of the surface area (27 m^2 for each species) with an approximate density of 100 *Eichhornia crassipes* plants per m^2 and 60 *Pistia stratiotes* plants per m^2 during the study period. One square meter of each plant was harvested once per month to prevent overgrowth.

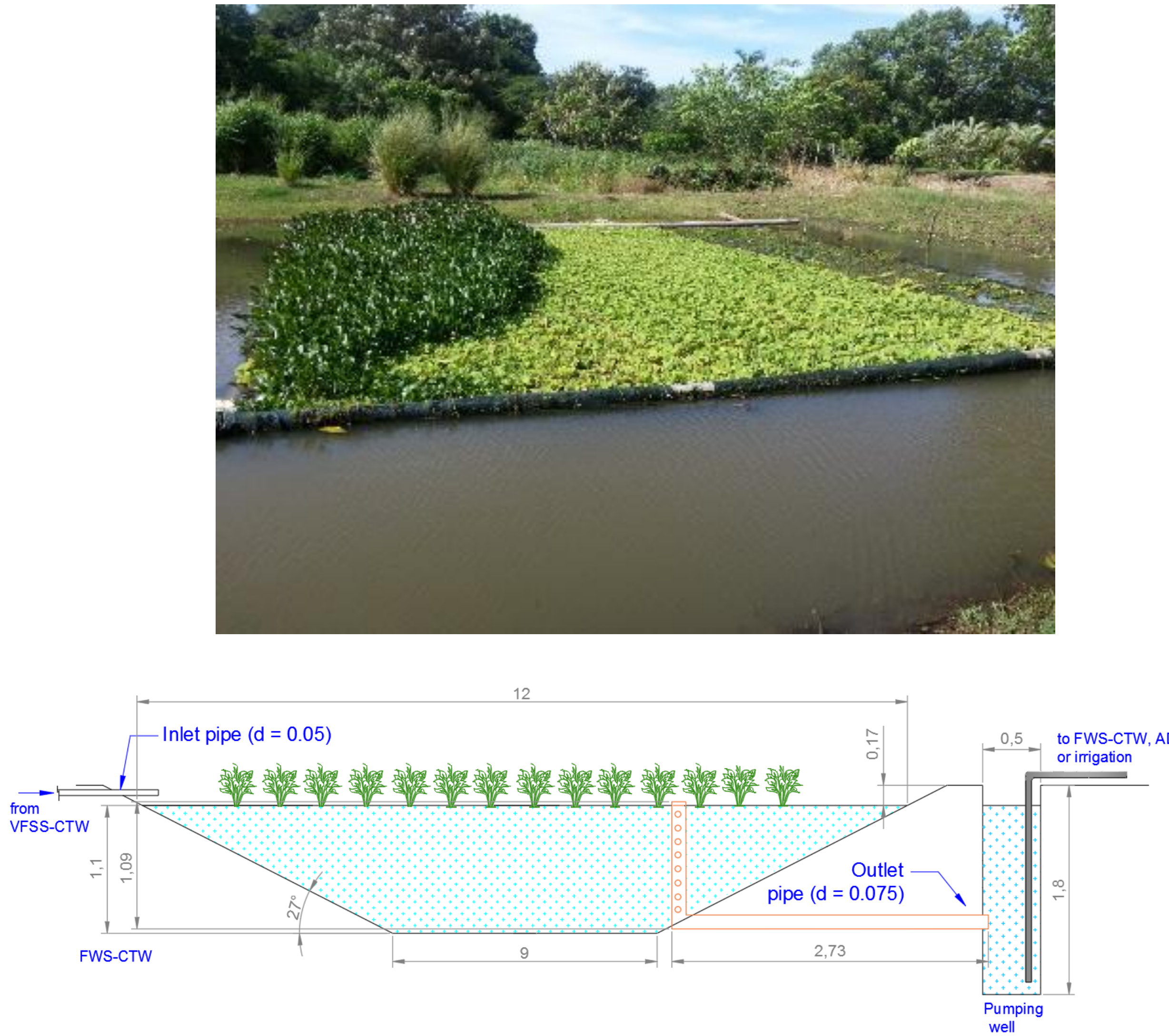

Figure 4. 3. Photograph and schematic diagram of FWS-CTW. Dimensions are in meters. Picture was taken in August 2015.

4.2.2. Hydrological balance

A hydrological balance was conducted for the HCTW (equation 4.2). Flows into and out of the VSSF-CTW and FWS-CTW were measured in m^3/week. Liquid digestate was discharged into the VSSF-CTW via gravity; time of flow per transfer was determined manually by recording the time it took the flow to fill a 1 m^3 tank. Flow transferred from the VSSF-CTW to the FWS-CTW and treated water pumped from the FWS-CTW was measured by recording the time each pump operated, as the flow rate of each pump was known from their specifications (VSSF-CTW pump = 110 L/minute, FWS-CTW pump = 150 L/minute). A weather station located on-site (IMN, 10.00 m N, -84.26 m W) recorded precipitation (mm), temperature (°C), relative humidity (%), wind speed (m/s), and solar radiation (MJ/m^2h) every two minutes. Meteorological data were analyzed on a weekly basis to measure precipitation and estimate evapotranspiration. Evapotranspiration was estimated using the FAO Penman-Monteith equation (Allen, Pereira et al. 1998). Runoff was calculated for the drainage area of 1,500 m^2, applying the curve number method (Ward and Trimble 2004). The volume of water stored (stored volume, SV, in m^3/week) in both the VSSF-CTW and the FWS-CTW was directly measured by recording the height of water in each pumping well on a daily basis, and averaging to find a weekly stored volume.

4.2.3. Treatment performance

Weekly samples were collected from August 2015 to March 2016. Liquid samples were collected from the digester effluent tank and effluents of both the VSSF-CTW and FWS-CTW. Samples were collected using 1 L bottles, which were capped with a lid and kept at 4°C until analysis, following certified methodologies for collection of samples established by the Water Quality Laboratory at the Research Center of Environmental Pollution (CICA-LCA), University of Costa Rica). CICA-LCA follows standard methodologies accredited by the Costa Rican Accreditation Institute (ECA). Measured water quality parameters were temperature, pH, COD, TS, volatile solids (VS), fixed solids (FS), TN, NO_3, NH_4, TP, and PO_4. Temperature and pH were measured using a pH meter (model HI-2211 from Hanna Instruments, UK). COD and solids, including VS and FS, were analyzed following Hach method #8000 and Hach method #8276, respectively. A DRB 200 reactor (Hach product #LTV082.53.40001) and a DR 900 multiparameter handheld colorimeter (Hach product #9385100) were used to digest and measure COD digestion vials (high range digestion vials from 0 to 1,500 mg COD/L, Hach kit). For TS, VS, and FS, samples were dried in disposable aluminum dishes (VWR®, catalog number 25433-008) for 24 hours in a StabilTherm gravity oven (model OV-12A from Blue M, East Troy, WI) at 100°C. Dishes were placed in a desiccator to cool for 30 minutes and were then weighed on an analytical balance (Ohaus Corporation, Mexico). Next, samples were placed in a StableTemp furnace (model CBFS516A from Cole-Parmer, Vernon Hills, IL) at 500°C for 30 minutes to determine FS and VS. TN (method MAQA-40), NO_3 (method MAQA-20), NH_4 (method MAQA-38), TP (method MAQA-1), and PO_4 (method MAQA-20) analyses were conducted at CICA-LCA. CICA-LCA analyses are based on the Standard Methods for the Examination of Water and Wastewater (Rice and Bridgewater 2012). Method 5310 B (modified combustion method) was used to assess TN (TOC-V CSH/CSN from Shimadzu, Columbia, MD); methods 4500-P D and 4500-NH_3 F were used to measure phosphorus and ammonium by UV-visual spectrophotometry (Evolution 600 from Thermo Scientific, Madison, WI); method 4110 B was followed for determining NO_3 and PO_4 by an ion chromatography (model MIC II from Metrohm, Switzerland).

The geotextile membrane was cleaned when needed and solids were collected and weighed by an industrial scale (Romanas Oconi S.A., Costa Rica). Moisture content of sediments collected by the geotextile membrane followed the previously described Hach method #8276. Carbon, nitrogen, and phosphorus concentrations of the sediments were analyzed at Agronomy Research Center (CIA) at the University of Costa Rica. Nitrogen and carbon concentrations were measured with Method SC09-LSF-P06 (Dunas method) using an autoanalyzer (Vario Cube from Elementar, Philadelphia, PA), while phosphorus concentrations were measured with method

SC09-LSF-P10 using an ICP-plasma atomic emission spectroscopy (Optima 8300 ICP-OES Spectrometer from Perkin Elmer, Spain).

The JMP®, Version 10.0.0 (SAS Institute Inc., Cary, NC) was used for statistical analyses. The Mann-Whitney test for two independent samples was conducted for determining significant differences between effluent concentrations in the HCTW for both low and high precipitation periods.

4.2.4. Modeling approach

The median weekly precipitation during the study period was 1.91 cm/week. Weeks with precipitation from 0.00 to 1.91 cm/week, occurring from January to March 2016, were considered to belong to the low precipitation period, while weeks with precipitations higher than 1.91 cm/week, occurring from August to December 2015, were considered to belong to the high precipitation period. More mathematically robust parameters that described the treatment performance of the VSSF-CTW and the FWS-CTW, during both low and high precipitation periods, were selected using PCA. Parameters included in the PCA were loading (g/m^2/week) and output concentrations (mg/L) of each water quality parameter analyzed, as well as, the influent flow rate (Q_i), precipitation ($P(A)$), and runoff (Q_c). Parameters that showed the high eigenvalues of principal components were identified as the key parameters; the cumulative percentage of these parameters accounted for close to 70% of the total variance. Once the key parameters were identified, graphical representations (loading charts) of those parameters were generated for each wetland and each precipitation period. Linear regression (R^2) was conducted for each graphical representation. Additionally, attempts to fit the data with contaminant removal models were made. Contaminant removal models were run on a spreadsheet and parameters were fitted using solver (Excel Microsoft) to minimize the sum of squares between actual and predicted outlet concentrations.

4.3. Results and discussion

4.3.1. Hydrological balance

Inputs (Q_i, Q_c and *P(A)*) and outputs (Q_o and *ET(A)*) of the hydrological balance for both the VSSF-CTW and FWS-CTW are shown in Table 4.2. The change in volume of water stored in each cell (D_s), calculated using the equation 4.2, and the average volume of water stored in each cell (SV) are shown in Table 4.2. Runoff from the surrounding drainage area entered the cells during October, November, and December 2015. *ET(A)* was higher during January to March than the *ET(A)* from September to December, due to high temperatures and dry conditions that are typical for January to April in Alajuela, Costa Rica. Consequently, there were periods of zero transfer of water from the VSSF-CTW to the FWS-CTW and from the FWS-CTW to the irrigations system during January to March 2016. Instead, effluent water was stored within the wetlands to support plant growth and was only discharged to provide 5 m^3 per week to the digester. For example, in the FWS-CTW, the hydrological balance was negative and consequently the stored volume of water substantially decreased from a daily average of 40 m^3 in January 2016 to 4.41 m^3 in March 2016.

Table 4. 2. Hydrological balance at the HCTW.

Month	Weather			VSSF-CTW				FWS-CTW			
	(m^3/d)										
	P(A)	ET(A)	Qc	Qi	Qo	Ds	SV	Qi	Qo	Ds	SV
Sept-2015	1.11 ±0.28	0.142 ±0.008	--	0.623± 0.058	0.244± 0.091	1.35± 0.44	20.6± 0.2	0.244± 0.091	0.259± 0.095	0.953± 0.474	87.2± 0.9
Oct-2015	2.34 ±0.85	0.110 ±0.099	6.76± 3.62	0.568± 0.045	5.64± 4.46	3.91± 9.07	37.4± 2.5	5.64± 4.46	7.91± 5.95	6.72± 14.90	115± 6
Nov-2015	1.97 ±0.74	0.135 ±0.033	6.95± 3.33	0.770± 0.001	8.85± 3.97	0.705± 8.074	46.7± 2.1	8.85± 3.97	12.3± 5.3	5.33± 13.40	119± 4
Dec-2015	0.618 ±0.663	0.233 ±0.025	1.83± 1.62	0.674± 0.091	0.417± 0.157	2.47± 2.55	21.6± 1.6	0.417± 0.157	0.823± 0.321	1.81± 2.78	94.0± 1.4
Jan-2016	0.004 ±0.005	0.285 ±0.018	--	0.639± 0.075	0.384± 0.152	-0.0256± 0.2500	6.15± 0.32	0.384± 0.152	0.886± 0.185	-0.783± 0.360	40.3± 4.0
Feb-2016	0.003 ±0.002	0.293 ±0.044	--	0.869± 0.060	0.120± 0.044	0.458± 0.150	5.77± 0.13	0.120± 0.044	0.206± 0.170	-0.376± 0.265	13.4± 0.6
Mar-2016	0.000	0.319 ±0.009	--	0.856± 0.001	0.0391± 0.0001	0.498± 0.010	6.25± 0.46	0.0391± 0.0001	0.384± 0.237	-0.664± 0.241	4.41± 4.94

a. Weather columns apply to both VSSF-CTW and FWS-CTW.

b. For each CTW, Q_i: influent flow, Q_o: effluent flow, *Ds*: change in volume of water stored in the CTW (a negative sign (-) indicates that the inputs are lower than the outputs), and *SV*: stored volume of water in the wetland (directly measured by assessing the height of water in each cell).

4.3.1.1. Hydrological balance on the VSSF-CTW

The VSSF-CTW has a water storage capacity in the media of 30.9 m^3, with an additional 80 m^3 of storage above the media. The wetland was flooded only during October and November 2015, when 6.5 m^3 and 15.8 m^3 of water per day was stored above the media, respectively (Figure 4.4). Under-sizing of diversion structures around the HCTW allowed runoff to enter the VSSF-CTW; however, the VSSF-CTW had sufficient storage capacity to hold both runoff and precipitation within its berms. During October and November, transfer of water from the VSSF-CTW to the FWS-CTW was increased to avoid spillover of untreated wastewater from the VSSF-CTW. The additional input of water due to precipitation and runoff diluted the applied wastewater (Figure 4.4.a). Negatively, runoff introduced uncertainty to the experimental results, as concentrations of pollutants in the runoff were not measured.

(a) (b)

Figure 4. 4. VSSF-CTW during the medium to high precipitation phase. a. A dilution effect could occur due to inputs such as precipitation and runoff. b. Berms had the capacity to hold water over the filter media and avoid spillover. Pictures were taken in October 2015.

Little precipitation occurred from January to March 2016. During this dry period, operation of the VSSF-CTW was modified to maintain saturated, anoxic conditions in the bottom of the cell by gradually decreasing the effluent flow from 1 m^3/d to 0.384 – 0.391 m^3/d. On average, the VSSF-CTW was 30% full, with saturated conditions in the bottom 0.20 m and unsaturated conditions in the top 0.90 m. HLRs (only considering wastewater) averaged 0.00474 ± 0.00032 m/d during the dry period and 0.00568 ± 0.00053 m/d during the rainy period. These values were substantially lower than published values which range from 0.0200 to 0.200 m/d (Kadlec and Wallace 2009). Wastewater hydraulic loading was low for two reasons. First, loading rates were determined from pollutant loading rates since influent concentrations of COD (>6,000 mg/L) and TS (>4,000) were very high. Pollutant loading rates varied from 24.7 – 37.2 g COD/m^2/d and from 16.5 – 24.8 g TS/m^2/d. These loads were higher than recommended loads for COD (< 20 g/m^2d) and TS (<5 g /m^2d) (Winter and Goetz 2003). Second, the Q_i applied to the VSSF-CTW was almost 50% less than the designed 1 m^3/d during the high precipitation period, whereas during the low precipitation period, the Q_i was 15% lesser than the designed when days that had no discharge were included in the average hydraulic loading rate. Discharge of liquid digestate in the wetland was conducted only from Monday to Friday and on some weekdays the liquid digestate was not discharged due to flooding in the wetland, geotextile cleaning, holidays, and days when equipment was broken in the system. Inclusion of precipitation and runoff in the hydraulic loading rate substantially increased the HLR; actual HLR during October and November 2015 was 0.0700 m/d, which was between the suggested ranges. HRTs, calculated as the stored volume divided by the total flow rate in ($P(A) + Q_c + Q_i$), ranged from 3.87 to 11.9 d during the rainy period and 6.69 to 9.55 d during the dry period. These HRTs were similar to the ones calculated by Avila, Salas et al. (2013) in a VSSF-CTW during a rainy period (3.5 d) and dry period (6.88 d) in Spain.

4.3.1.2. Hydrological balance on the FWS-CTW

The FWS-CTW has a storage capacity of 152 m^3. During the high precipitation period, additional water was discharged from the FWS-CTW to avoid overtopping the VSSF-CTW cell after large runoff events. On average, 7.91 and 12.3 m^3/d were discharged during October and November 2015, respectively, as compared to an average of 0.607 m^3/d during the dry period. As previously mentioned, runoff could impact modeling as the concentrations of pollutants in the runoff were unknown and the volume of runoff was approximated. During the dry period, the FWS-CTW had sufficient capacity to store water from the VSSF-CTW; the stored value decreased with time due to higher Q_o than Q_i, as well as higher $ET(A)$ (Table 4.2). The Q_o was higher that the Q_i because the effluent from the FWS-CTW was used to feed the digester at a rate of 5 m^3/week. This rate was higher than the Q_o from the VSSF-CTW in order to keep water at the bottom of the VSSF-CTW.

The FWS-CTW had an HLR of 7.24×10^{-3} m/d, which is within the recommended range of 7.00×10^{-3} to 5.00×10^{-2} m/d (Kadlec and Wallace 2009). The HRT of the FWS-CTW varied substantially from 6.72 to 64.4 d during the period of high precipitation and from 103 to 112 d during the period of low precipitation. Lower HRTs during the high precipitation period were due to high flow rates (measured as $P(A) + Q_c + Q_i$), especially during October and November 2015. Based on HLR suggested by Kadlec and Wallace (2009), recommended HRTs for FWS-CTWs range from 22 to 157 d (HLR = 7.00×10^{-3} m/d). The HRT of our FWS-CTW was within this range during September (64.4 d), December (32.8 d), and from January to March 2016 (>100 d). HRT was shorter than desired for October (7.84 d) and November (6.72 d).

4.3.2. Treatment performance

Table 4.3 summarizes the treatment performance of the VSSF-CTW and FWS-CTW. The effluent from the system is compared to the Costa Rican water quality standards (COD < 150 mg/L, TS < 50 mg/L, TN < 50 mg/L, TP < 8 mg/L, 5 < pH < 9, and 15 °C < T < 40 °C) for discharging treated effluents into bodies of water or onto crop fields (MINAE-MSP 2007).

Table 4. 3. Characteristics of studied parameters at the effluent tank, the VSSF-CTW, and the FWS-CTW for both high (n =16) and low (n = 9) precipitation periods.

	Parameter	**Effluent Tank**	**VSSF-CTW**		**FWS-CTW**		**HCTW**
		Mean ± S.E.	Mean ± S.E.	% Removal	Mean ± S.E.	% Removal	% Removal
High precipitation period	COD (mg/L)	6,841 ± 681	66.5 ± 12.8	99.0 ± 0.3	34.3 ± 2.05	48.4 ± 0.1	99.5 ± 0.1
	TS (mg/L)	4,399 ± 383	543 ± 63.5	87.6 ± 0.2	243 ± 23.7	55.1 ± 0.2	94.4 ± 0.2
	VS (mg/L)	1,462 ± 210	171 ± 35.4	88.3 ± 0.3	92.5 ± 19.2	45.9 ± 0.2	93.7 ± 0.4
	FS (mg/L)	2,938 ± 267	372 ± 47.5	87.3 ± 0.2	151 ± 13.7	59.4 ± 0.1	94.8 ± 0.2
	NH_4 (mg/L)	1,312 ± 119	13.9 ± 3.38	98.9 ± 0.3	1.20 ± 0.332	91.4 ± 0.5	99.9 ± 0.4
	NO_3 (mg/L)	3.64 ± 0.550	71.4 ± 22.8	(1800 ± 8)	1.23 ± 0.320	98.3 ± 0.6	66.2 ± 0.3
	TN (mg/L)	1,078 ± 101	34.0 ± 8.55	96.8 ± 0.3	2.03 ± 0.458	94.0 ± 0.4	99.8 ± 0.3
	PO_4 (mg/L)	80.0 ± 48.6	1.36 ± 0.181	98.3 ± 0.7	0.600 ± n.d.	55.9 ± 0.1	99.2 ± 0.6
	TP (mg/L)	106 ± 16.3	0.801 ± 0.131	99.2 ± 0.3	0.913 ± 0.145	(14.0 ± 0.1)	99.1 ± 0.3
	pH	7.95 ± 0.0434	6.93 ± 0.102	--	6.63 ± 0.119	--	--
	Temperature (°C)	26.4 ± 0.530	25.7 ± 0.550	--	25.4 ± 0.452	--	--
Low precipitation period	COD (mg/L)	8,095 ± 914	564 ± 120	93.0 ± 0.3	288 ± 57.9	48.8 ± 0.2	96.4 ± 0.3
	TS (mg/L)	6,630 ± 853	2,926 ± 272	55.8 ± 0.1	892 ± 125	69.5 ± 0.2	86.5 ± 0.2
	VS (mg/L)	2,439 ± 589	1,087 ± 155	55.4 ± 0.2	288 ± 53.1	73.5 ± 0.2	88.2 ± 0.4
	FS (mg/L)	4,191 ± 340	1,839 ± 172	56.1 ± 0.1	604 ± 87.4	67.1 ± 0.2	85.6 ± 0.2
	NH_4 (mg/L)	1,714 ± 99	78.0 ± 30.9	95.4 ± 0.4	15.4 ± 6.79	80.3 ± 0.7	99.1 ± 0.5
	NO_3 (mg/L)	12.7 ± 6.23	377 ± 121	(2,800 ± 9)	22.4 ± 10.7	94.1 ± 0.7	(76.1 ± 0.4)
	TN (mg/L)	1,443 ± 108	246 ± 28.2	82.9 ± 0.2	15.7 ± 6.79	93.6 ± 0.2	98.9 ± 0.3
	PO_4 (mg/L)	8.50 ± 2.6	3.74 ± 0.876	56.0 ± 0.3	1.92 ± 0.268	48.7 ± 0.2	77.4 ± 0.4
	TP (mg/L)	156 ± 34	1.88 ± 0.756	98.8 ± 0.6	0.632 ± 0.0845	66.4 ± 0.4	99.6 ± 0.3
	pH	8.35 ± 0.031	7.59 ± 0.0936	--	7.63 ± 0.0551	--	--
	Temperature (°C)	26.8 ± 0.29	26.8 ± 0.242	--	26.6 ± 0.224	--	--

*. Mann-Whitney test for two independent samples comparing the HCTW effluent during both low and high precipitation periods (alpha = 0.05). Removal rates were significantly different between the high and low precipitations periods (p<0.05, with the exception of total phosphorus (p = 0.208).

During the high precipitation period, removal for all water quality parameters except NO_3 (66.2%) were higher than 90% for the HCTW (Table 4.3). After the VSSF-CTW, water quality parameters such as COD, TN, TP, pH, and temperature satisfied discharge standards for Costa Rica. Further treatment at the FWS-CTW reduced the concentration of these parameters. Despite the substantial reduction of TS, 87.6% after the VSSF-CTW and 94.4% after the HCTW, the effluent did not meet water discharge standards. Despite this, the effluent from the HCTW was considered reclaimed for four reasons. For irrigation, Avila, Salas et al. (2013) indicated that TS of mineral origin cause a minimal impact if is discharged onto land or into aquatic ecosystems. In our case, the effluent from the HCTW had low COD concentration and low VS to TS ratio (38.0 ± 5.1%), and the sediments in the effluent were considered of mineral origin. Secondly, regulations in Costa Rica allows a discharge of wastewater from livestock activities with concentrations of 200 mg TS/L (MINAE-MSP 2007), and this concentration was met by the effluent of the HCTW if considered only the FS (151 ± 13.7 mg/L). Third, land application of liquid manures is a common practice in Costa Rica. Liquid manures are defined to have less than 4% TS, with typical measured concentrations up to 23.9 g TS/L (Lorimor, Powers et al. 2004). Thus, TS concentrations were not considered a barrier to use of the effluent for irrigation, as TS concentration in the liquid manures is 44 times larger than the concentration from the VSSF-CTW effluent in this study. Additionally, the sequence of recirculation and discharge likely increased TS in the effluent of the FWS-CTW; water within the FWS-CTW was recirculated from 2 – 4 am and effluent samples were collected prior to 8 am. It is expected that modifying this schedule would substantially decrease the TS in the effluent of FWS-CTW; however, this action was not immediately taken during this study so that effluent concentrations during the rainy and dry periods could be directly compared. Finally, the effluent from the HCTW can be reused for dilution of feed for the digester.

During the low precipitation period, total removal for the HCTW varied from 77.4 to 99.6% (Table 4.3). NO_3 effluent concentrations from the HCTW increased with respect to initial concentrations in the effluent tank. After the VSSF-CTW, only TP, pH, and temperature satisfied discharge standards for Costa Rica, whereas the FWS-CTW further removed TN to a permitted concentration. In contrast, COD and TS were above the maximum discharge standard after both the VSSF-CTW and the FWS-CTW. During this period, the water from the FWS-CTW was used only for dilution of feed for the digester.

Thus, there was a difference in the treatment performance during the two precipitation periods. During this study, lower concentrations were obtained during the high precipitation period (Table 4.3). The Mann-Whitney test for two independent samples indicated that there were significant differences ($p < 0.05$) in the HCTW effluent concentrations for each water quality parameter, except TP ($p = 0.208$), between the low and high precipitation periods (Table 4.3). Dilution, due to precipitation, and concentration, due to evapotranspiration, likely contributed to differences in HCTW effluent concentrations between the low and precipitation periods. Avila, Salas et al. (2013) indicated that inlet concentrations for a VSSF-CTW were lower during a wet period (59.0 mg/L) than during a dry period (294 mg/L), due to the dilution effect caused by precipitation. Consequently, outlet COD concentrations were lower during a wet period (29 mg/L) than during a dry period (46 mg/L) (Avila, Salas et al. 2013). Additionally, in this study, it was likely that concentration of pollutants occurred from January to March 2016. Low HLR, scarce precipitation events, and high temperatures substantially decreased the water levels in the HCTW, thus increasing the concentration of pollutants. For example, the water at the FWS-CTW was green (Figure 4.5), likely due to algae bloom promoted by high temperatures and availability of nutrients (e.g., nitrate and phosphate). Availability of nutrients occurred due to resuspension of sediments by the movement of the water during recirculation of water in the FWS-CTW and during sampling and plant harvest. It was likely that these conditions caused high concentrations during the low precipitation period in the HCTW.

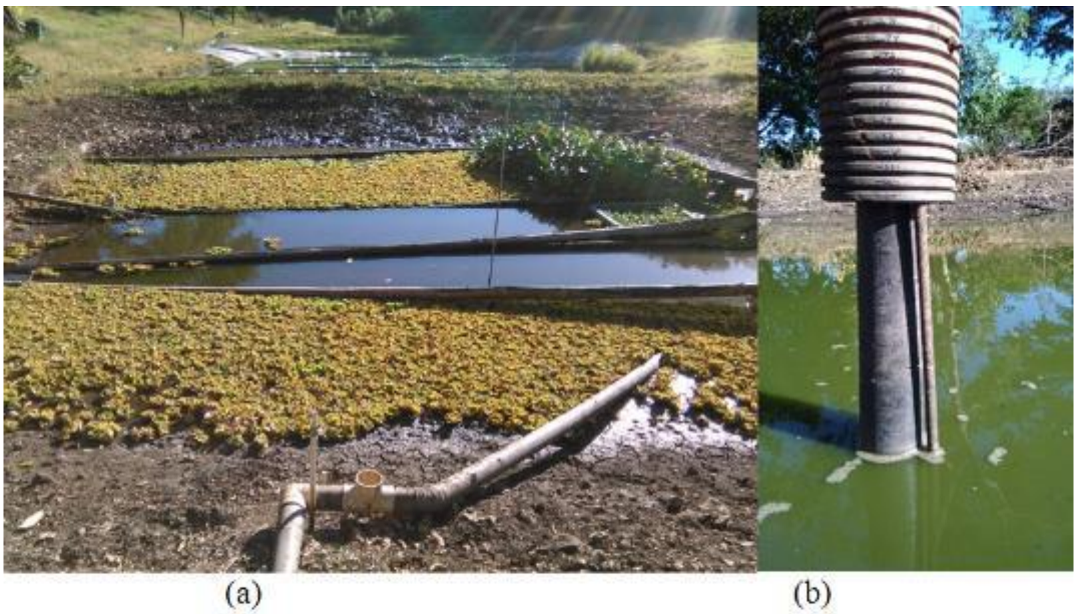

Figure 4. 5. FWS-CTW during the low precipitation period. a. Low HLR, scarce precipitation events, and high temperatures substantially decreased the water at the FWS-CTW. b. Growth of algae bloom. Pictures were taken in February 2016.

Use of multiple types of CTWs in-series aims to increase removal by balancing the strengths and weakness of each type of CTW and is a common approach to achieve high removals of pollutants (Tunçsiper 2009, Avila, Salas et al. 2013, Comino, Riggio et al. 2013, Vymazal and Kröpfelová 2015, Avila, García et al. 2016). In this case, a VSSF-CTW was combined in series with a FWS-CTW. The VSSF-CTW was placed first because the influent had high concentrations of NH_4 (>1,000 mg/L), COD (>6,000 mg/L), and TS (>4,000 mg/L). The aerobic conditions that are predominant in VSSF-CTW facilitate nitrification (Vymazal and Kröpfelová 2015). Then, the FWS-CTW, which is predominantly anaerobic, supports denitrification (Vymazal 2007, Kadlec and Wallace 2009). In our VSSF-CTW, concentrations of NH_4 decreased by more than 95.0% while NO_3 concentrations increased substantially. NO_3 accounted for less than the 0.120% of the TN in the liquid digestate and 34.6 to 47.6% of the TN in the effluent from the VSSF-CTW. This increase in NO_3 concentrations indicated that nitrification was a major process responsible for ammonium removal in the VSSF-CTW. Additionally, >80% of TN in the liquid digestate was removed by the VSSF-CTW and removal of ammonium was much greater than the net production of nitrate. In contrast, nitrogen removal in a sand-based VSSF-CTW planted with *T. angustifolia L.* and *Cyperus alternifolius L* in Thailand was limited due to a lack of anoxic conditions and subsequent denitrification (Kantawanichkul, Kladprasert et al. 2009, Avila, Salas et al. 2013). As our VSSF-CTW was always at least 30% full, with saturated conditions in the bottom 0.2 m and unsaturated conditions in the top 0.90 m, it is likely that nitrification did occur. Additional processes that could have contributed to nitrogen removal, aside from denitrification and nitrification, include microbial and plant assimilation, anaerobic ammonium oxidation (anammox), and volatilization of ammonium (Kantawanichkul, Kladprasert et al. 2009, Avila, Salas et al. 2013). However, neither volatilization nor anammox were expected to occur at high rates as the water pH and temperature were 6.93 and 25.7°C, respectively, which are not conducive to either process (Lee, Fletcher et al. 2009, He, Tao et al. 2012). Furthermore, plant uptake is typically assumed to represent no more than 10% of removal (Vymazal 2007, Vymazal 2011), Additionally, some removal of TN might occur from filtration of solids by the geotextile placed at the inlet area of the VSSF-CTW, which is analyzed in the next section.

NO_3 concentrations decreased by more than 90% in the FWS-CTW (Table 4.3), indicating that denitrification was likely a major nitrogen removal process in the FWS-CTW (Kantawanichkul, Kladprasert et al. 2009, Avila, Salas et al. 2013). Plant uptake also likely contributed to nitrogen removal. The FWS-CTW was important for removal of TN. Table 4.3 shows that percentage removal for almost all of the water quality parameters was low, from 45 to 75%, but higher for NH_4, NO_3, and TN (higher than 80%), during both low and high precipitation periods.

Despite the high OLR reported in this study (24.7 – 37.2 g COD/m^2/d and 16.5 – 24.8 g TS/m^2/d), the HCTW performed similarly to other wetlands. A VSSF-CTW receiving an organic loading of 19.6 g BOD/m^2/d achieved removal of 92% for BOD, 89% for COD, 95% for TS, 77% for TN, and 62% for TP (Tsihrintzis, Akratos et al. 2007). In our study, the FWS-CTW served as a polishing step and achieved lower treatment performance than Katsenovich, Hummel-Batista et al. (2009) and Avila, Salas et al. (2013), who demonstrated that adequate treatment by FWS-CTW can occur for higher loads. For example, in El Salvador, Katsenovich, Hummel-Batista et al. (2009) evaluated a FWS-CTW recieving municipal wastewater. Despite the higher HLR (0.206 m/d), a FWS-CTW planted with *Thypa angustifola* removed 81% for BOD (20 mg/L), 65% for COD (73 mg/L), 59% for TN (6.1 mg/L), and 67% for TP (1.9 mg/L) (Katsenovich, Hummel-Batista et al. 2009). Better performance was obtained by Avila, Salas et al. (2013); a FWS-CTW receiving municipal wastewater removed 98% of BOD (7.0 mg/L), 91% of COD (50 mg/L), 98% of TS (6.0 mg/L), 85% of TN (7.9 mg/L), and 34% for TP (5.3 mg/L) (Avila, Salas et al. 2013). Note that comparisons between systems should be taken with caution due to different configurations of the cells.

4.3.2.1. The geotextile membrane at the VSSF-CTW

The geotextile membrane (2 m x 2 m) was placed in a shallow trench of approximately 0.25 m depth, with the purpose to remove suspended solids still present in the liquid digestate (Figure 4.6). This area has sufficient volume to hold the one cubic meter discharged each day. Ponded in the geotextile membrane area, the liquid digestate slowly passed through the membrane and infiltrated only into the filter media beneath the membrane (4 m^2 of the total median treatment area of 138 m^2). The geotextile was cleaned three times during this study (Table 4.4). The first batch was collected before heavy rains started. The dry conditions during the day allowed the sediments to lose moisture and be easily collected. In contrast, in October and November 2015, it was not possible to collect sediments as the wetland was flooded (Figure 4.4). The last two batches were collected after rain events from October to November 2015 (second batch) and after intermittent rains during January 2016 (third batch). Even though the second batch was after 9 weeks of operation, during which rains and flooding occurred in the VSSF-CTW, more sediments were collected after 5 weeks of operation in the third batch. Due to flooding, it was likely that a portion of the sediments, during the second batch, were dispersed and settled in in the main treatment area, rather than being isolated to the geotextile area (Figure 4.4). Additionally, solids had accumulated in the effluent tank to the height of the outlet pipe by January 2016, increasing the discharge of solids into the VSSF-CTW. For these two reasons, there was a substantial difference in the mass of dry sediments collected during the first and third collection. Consequently, the effluent tank was cleaned out in January 2016.

Table 4. 4. Sediments collected from the geotextile membrane at the VSSF-CTW.

Period	Number of weeks	Collected sediments	Moisture content	Collected sediments
		(wet, kg)	(%)	(dry, kg /m2/week)
08/31/2015 to 10/06/2015	5	19.2 ± 0.5	57.3 ± 0.5	0.41 ± 0.01
10/06/2015 to 12/07/2015	9	399 ± 1	90.66 ± 0.4	1.04 ± 0.01
12/07/2015 to 01/12/2016	5	855 ± 1	90.66 ± 0.6	3.99 ± 0.03

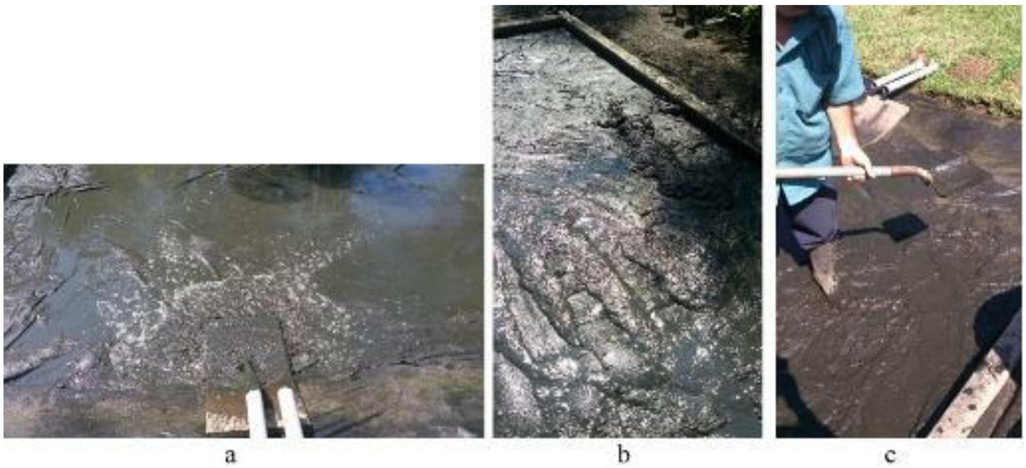

Figure 4. 6. Geotextile membrane placed in the VSSF-CTW inlet area. a. Liquid digestate was discharged by a PVC pipe (diameter = 0.05 m) into the geotextile membrane. b. Solids were accumulated on the geotextile while the liquid passed through the membrane and infiltrated in the filter media of the VSSF-CTW. c. A worker cleaning up the geotextile membrane. Pictures were taken in October 2014.

In terms of mass removal, the geotextile contributed to performance of the VSSF-CTW. Table 4.5 shows the mass and removal loading rates for the geotextile and the overall VSSF-CTW for COD (used as a proxy for carbon), TS, TN, and TP. In general terms, the geotextile membrane accounted for a small portion of the total removal in the VSSF-CTW. Average mass removal by the geotextile membrane was 4.81% for COD, 27.9% for TS, 13.4% for TN, and 19.5% for TP. The geotextile membrane prevented direct contact of these sediments with the substrate media (coarse sand), likely reducing clogging, as formation of a crust was not observed in the treatment area of the VSSF-CTW. Additionally, accumulation of sediments in the geotextile membrane facilitated the collection of this material for further treatment (composting).

In this way, the geotextile membrane decreased the mass load to be treated by the VSSF-CTW, reduced potential clogging, and allowed recovery of nutrients. Many authors and CTW guidelines suggest a pretreatment system before discharging wastewater into the wetland (Garcia-Perez, Harrison et al. 2011, Comino, Riggio et al. 2013, de la Varga, Díaz et al. 2013); however, the implementation of a geotextile membrane in the inlet area provided a low-cost, low-maintenance approach to improving the performance of the VSSF-CTW.

Table 4. 5. Mass removal from the geotextile membrane and from the VSSF-CTW.

	Period[a]	Mass loading into the geotextile[b]	Mass removal by the geotextile[c]	Removal by the geotextile	Mass loading after the geotextile	Mass after the VSSF	Mass removal by the VSSF-CTW	Removal by the VSSF
		kg /m²/week		(%)	kg /m²/week			(%)
COD	1	4.32 ± 00502	0.109	2.52	4.21 ± 0.502	(6.99 ± 3.17) x10^{-4}	4.21 ± 0.502	99.9 ± 0.1
	2	10.1 ± 1.39	0.274	2.71	9.81 ± 1.39	0.0113 ± 0.00423	9.80 ± 1.39	99.9 ± 0.1
	3	11.2 ± 2.22	1.06	9.46	10.1 ± 2.22	0.110 ± 0.0775	9.93 ± 2.16	97.9 ± 0.7
	Avg.	8.92 ± 1.08	0.429 ± 0.0871	4.81 ± 0.01	8.49 ± 1.05	0.0558 ± 0.0268	8.44 ± 1.04	99.4 ± 0.3
TS	1	3.66 ± 0.488	0.410	11.2	3.26 ± 0.489	(7.80 ± 2.96) x10^{-3}	3.24 ± 0.486	99.8 ± 0.1
	2	6.41 ± 0.722	1.03	16.1	5.38 ± 0.772	(9.74 ± 3.60) x10^{-2}	5.28 ± 0.760	98.2 ± 0.6
	3	6.58 ± 2.30	3.99	60.6	2.59 ± 2.31	(4.43 ± 2.09) x10^{-2}	2.55 ± 2.29	99.9 ± 0.5
	Avg.	5.77 ± 0.734	1.61 ± 0.328	27.9 ± 0.1	4.15 ± 0.737	(6.17 ± 2.06) x10^{-2}	4.09 ± 0.727	99.0 ± 0.4
TN	1	0.706 ± 0.0906	0.0122	1.73	0.693 ± 0.0906	(3.61 ± 1.20) x10^{-4}	0.693 ± 0.0905	99.9 ± 0.1
	2	1.53 ± 0.199	0.0307	2.00	1.50 ± 0.199	(2.41 ± 0.61) x10^{-3}	1.49 ± 0.198	99.8 ± 0.1
	3	1.81 ± 0.156	0.118	6.52	1.69 ± 0.156	(4.29 ± 1.88) x10^{-3}	1.69 ± 0.156	99.7 ± 0.1
	Avg.	1.39 ± 0.145	0.186 ± 0.0298	13.4 ± 0.1	1.20 ± 0.136	(2.37 ± 0.63) x10^{-3}	1.20 ± 0.136	99.8 ± 0.1
TP	1	0.0567 ± 0.00924	0.00734	12.9	0.0493 ± 0.00924	(6.20 ± 2.88) x10^{-6}	0.0493 ± 0.00924	99.9 ± 0.1
	2	0.165 ± 0.0321	0.0185	11.2	0.147 ± 0.0321	(3.32 ± 1.81) x10^{-4}	0.146 ± 0.0319	99.7 ± 0.1
	3	0.205 ± 0.0729	0.0714	34.8	0.133 ± 0.0729	(1.93 ± 1.12) x10^{-5}	0.133 ± 0.0729	99.9 ± 0.0
	Avg.	0.148 ± 0.0269	0.0289 ± 0.00587	19.5 ± 0.1	0.119 ± 0.0254	(1.72 ± 0.978) x10^{-4}	0.119 ± 0.0254	99.8 ± 0.1

a. Period 1 from 08/31/2015 to 10/06/2015, period 2 from 10/06/2015 to 12/07/2015, and period 3 from 12/07/2015 to 01/12/2016.

b. Formula: Q_i*C_{in} /geotextile area/number of weeks.

c. Sediments collected times the amount of carbo (26.5 ± 2.66%), nitrogen (2.97 ± 0.307%), or phosphorus (1.79 ± 0.275%).

Abbreviation: average (Avg).

4.3.3. Modeling approach

PCA was conducted to statistically identify key parameters that described the treatment performance of the VSSF-CTW and the FWS-CTW, during both low and high precipitation periods. In general, parameters such as influent flow rate (Q_i), precipitation ($P(A)$), and runoff (Q_c) had low eigenvalues, and these parameters were not directly considered for modeling. Based on the eigenvalues extracted from the loading matrix principal components, the more mathematically robust parameters that described the treatment performance were the water quality parameters (Table 4.6). During the low precipitation period, the treatment performance at the VSSF-CTW was explained by TS load, COD out, and TN load, with a cumulative percentage of 76.3%. During the high precipitation period, COD load and COD out explained the treatment performance with a cumulative percentage of 67.2%. For the FWS-CTW, the treatment performance during the low precipitation period was explained by FS load, TS out, and TP load with a cumulative percentage of 74.1%; whereas, TS load, TS out, and NH_4 out explained the treatment performance during the medium high precipitation period with a cumulative percentage of 77.6%. Loading charts and contaminant removal models were evaluated for their ability to describe the treatment performance of selected water quality parameters.

Table 4. 6. Key water quality parameters for the HCTW during both high and low precipitation periods.

Period	VSSF-CTW				FWS-CTW			
	PC	Parameter	Eigenvalues	Cumulative percentage (%)	PC	Parameter	Eigenvalues	Cumulative percentage (%)
High precipitation period (n=16)	1	COD load	0.975	37.8	1	TS load	0.994	42.2
	2	COD out	0.800	29.4	2	TS out	0.936	21.8
	3	--	--	--	3	NH_4 out	0.689	13.6
Low precipitation period (n=9)	1	TS load	0.913	37.8	1	FS load	0.972	34.0
	2	COD out	0.849	24.6	2	TS out	0.934	28.2
	3	TN load	0.733	13.9	3	TP load	0.773	11.9

The measured output concentrations (mg/L) were not correlated to input loading rates (g/m^2/week)) for the key water quality parameters, with R^2 values ranging from 0.0115 to 0.381(Figures 4.7 and 4.8). These poor relationships are normal for full-scale systems (Knight, Payne Jr et al. 2000). The internal variability in the wetlands (plant communities, internal hydraulics, and weather conditions) likely impacted treatment performance, and more time evaluating the wetlands could narrow the central tendency of the treatment performance for the HCTW in this study. Variability in treatment performance is expected, especially when large data sets are used for representing loading charts. Knight, Payne Jr et al. (2000) analyzed the Livestock Wastewater Treatment Database (LWDB) for North America. LWDB gathered information from 68 sites, 278 individual treatment cells, and more than 1,390 indivdual records with data for multiple parameters. Based on this data, Knight, Payne Jr et al. (2000) recommended against the use of loading charts (output concentrations vs. input loading rates) due to the poor fit of data (low R^2 values) for BOD (R^2=0.74) and TSS (R^2=0.30); these loading charts should be used with caution for NH_4 (R^2=0.87), TN (R^2=0.81), TP (R^2=0.70), and COD (R^2=0.89). In addition, Kadlec and Wallace (2009) indicated that loading charts are most usefull when inlet concentrations are constant; in contrast, inlet concentrations were highly variable in this study.

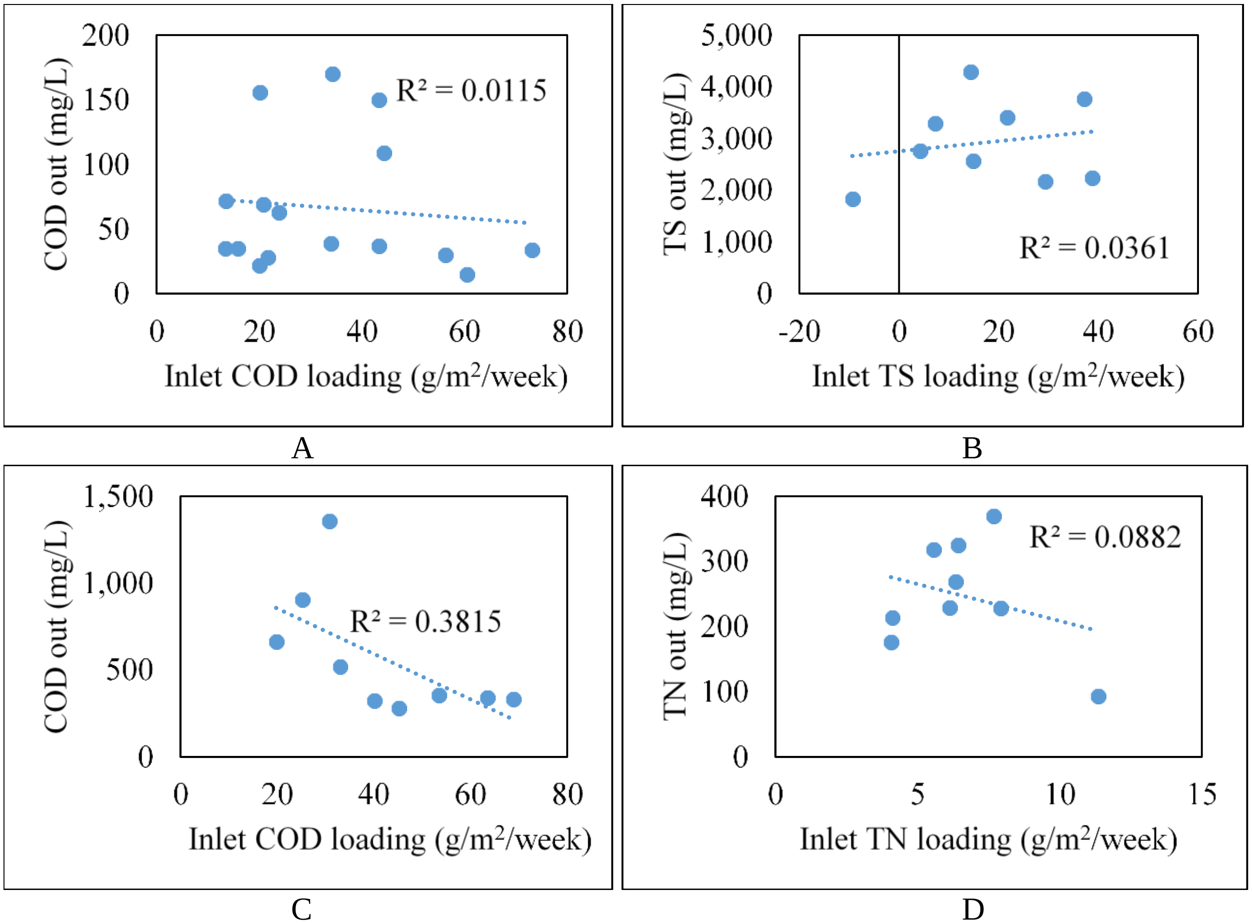

Figure 4. 7. Relationship between mass loading and outlet concentration at the VSSF-CTW for A) COD during the high precipitation period, B) TS during the low precipitation period, C) COD during the low precipitation period, and D) TN during the low precipitation period.

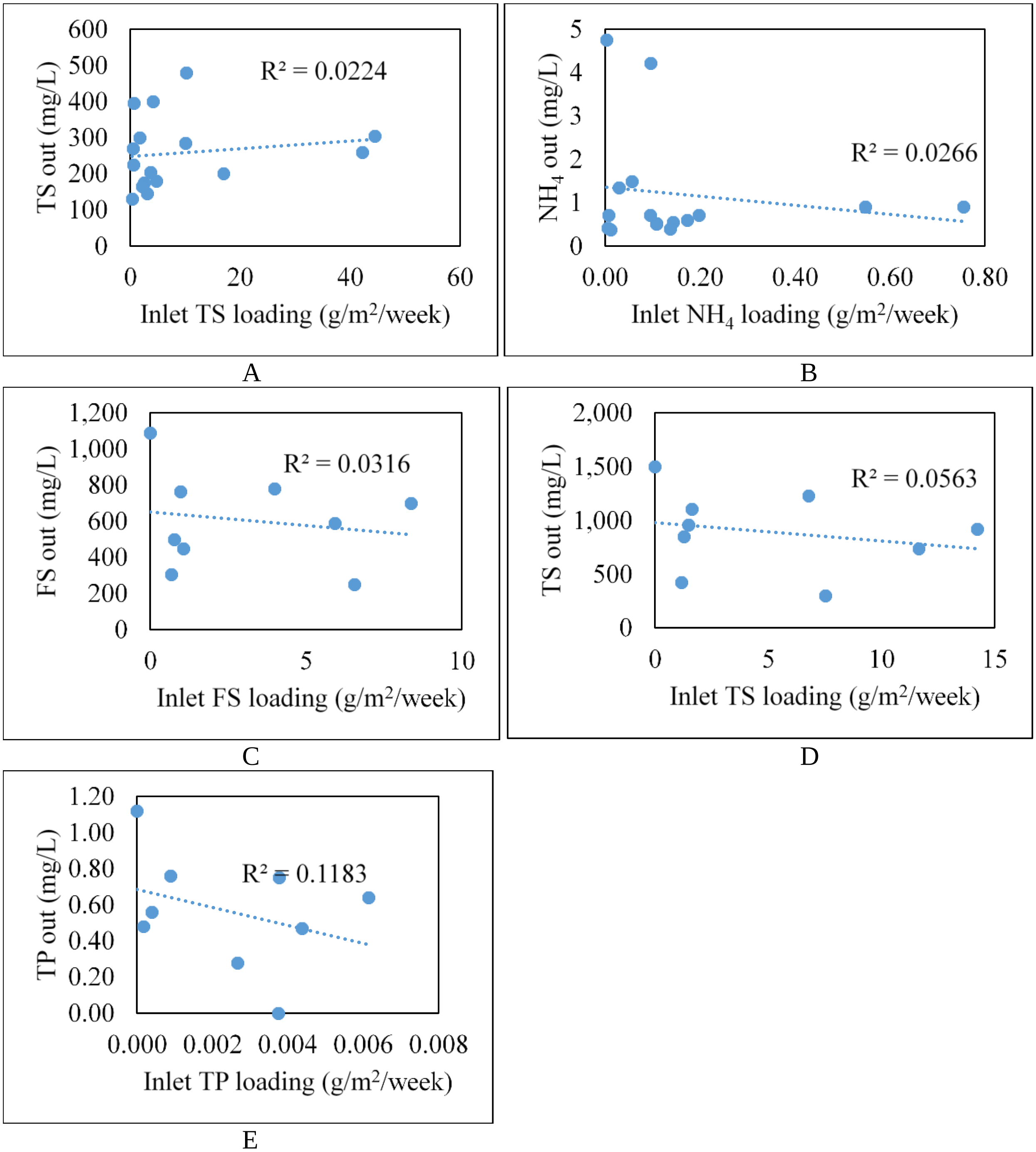

Figure 4. 8. Relationship between mass loading and outlet concentration at the FWS-CTW for A) TS during the high precipitation period, B) NH_4 during the high precipitation period, C) FS during the low precipitation period, D) TS during the low precipitation period, and E) TP during the low precipitation period.

Attempts to model the data with contaminant removal models were largely unsuccessful. Model parameters, such as K_{20} and θ_{20} for the plug flow model, K_{20}, θ_{20}, and C^* for the modified k-C^* model, P, k, and C^* for the TIS (p k-C^*) model, and k_0 and b for the TDR model, were fitted, based on the measured inlet and outlet concentration obtained during this study (Tables 4.7 and 4.8). For the VSSF-CTW, the sum of squares between actual and predicted outlet concentrations and low R^2 indicated that the fitted parameters should not be used for design purposes. Similar poor fits were found for the FWS-CTW, with the exception of modeling ammonium removal with the TIS model during the high precipitation period (sum of the squares = 17.3 and R^2 = 0.512). However; the fitted K_{20} and θ_{20} values were unreasonable compared to estimated values by Tanaka and Tanaka (2011). In general, it was expected to obtain fitted k-values for COD that were higher than literature values. Due to the temperature dependence of this rate, higher removal rates were expected in this study, as the HCTW is located in the tropics. The steady warmer climate across the year in the tropics is expected to yield constant removal rate, which allows the reduction in the required area for wetland treatment at the tropics (Zurita, De Anda et al. 2009). However, due to the poor fitting obtained in this study, the adjusted values cannot be considered for designing wetlands.

The background concentration (C^*) is considered in the modified k-C^* and TIS (P k-C^*) models. Temperature plays a role in the background concentration because microbes' degradation increases or decrease based on the temperature. In this study, C^* was initially estimated by equations or values indicated in the literature. The COD background concentration at subsurface CTW was estimated by equation 4.8. Similarly, TS at the VSSF-CTW ($C^* = 7.8 + 0.063C_i$) and at the FWS-CTW ($C^* = 5.1 + 0.16C_i$) were based on Kadlec's background equation concentration (Kadlec and Wallace 2009). In these cases, the C^* substantially varied due to the unsteady influent concentration for COD and TS (Table 4.7 and 4.8). Attempts to fit the value for C* for the key parameters in each wetland from the data were unsuccessful. Finally, the TDR model (Shepherd, Tchobanoglous et al. 2001) replaces C^* for two parameters (K_0 and b). Shepherd, Tchobanoglous et al. (2001) estimated k_0 values from 9 to 12 d^{-1} and b values from 2 to 5 d^{-1} for modeling COD removal in a subsurface flow constructed wetland for winery wastewater treatment. Attempts to fit k_0 and b to the data were also unsuccessful, yielding high sums of squares and low R^2 values between the predicted and measured effluent concentrations (Table 4.7). Shepherd, Tchobanoglous et al. (2001) clearly indicated that these parameters were obtained for a pilot-scale system and the validation at full-scale has not been done.

Table 4. 7. Contaminant removal models approach for the VSSF-CTW.

VSSF-CTW						
	Water quality parameter	**Contaminant removal models**				
		Model parameters	**Plug-flow**	**Modified k–C***	**TIS (P k–C*)**	**TDR**
			Adjusted value	**Adjusted value**	**Adjusted value**	**Adjusted value**
High precipitation period	**COD**	K_{20}, d^{-1}	1.29	10,528,454	--	--
		θ_{20}	2.91	56,908,317	--	--
		Area, m^2	141	84,060	77,945	167
		Depth, m	0.750	16,544,712	0.700	14.0
		C*, mg/L	--	--	--	--
		k, m/d	--	--	53,687,091	--
		P	--	--	3.00	--
		k_0, d^{-1}	--	--	--	9.49
		b, d^{-1}	--	--	--	1.65
		Sum of the square	108,6711	1,742,815	1,742,815	59,233
		R^2	0.0707	0.0551	0.0551	0.0137

Table 4. 7. (cont'd)

VSSF-CTW						
	Water quality parameter	**Contaminant removal models**				
		Model parameters	**Plug-flow**	**Modified k–C***	**TIS (P k–C*)**	**TDR**
			Adjusted value	**Adjusted value**	**Adjusted value**	**Adjusted value**
Low precipitation period	**COD**	K_{20}, d^{-1}	0.890	0.540	--	--
		θ_{20}	1.91	1.61	--	--
		Area, m^2	9.26	37.7	138	10,331
		Depth, m	0.480	0.420	0.700	1,025
		C*, mg/L	--	--	--	--
		k, m/d	--	--	0.200	--
		P	--	--	3.00	--
		k_0, d^{-1}	--	--	--	248
		b, d^{-1}	--	--	--	1,601
		Sum of the square	1,938,972	1,474,965	1,802,721	1,838,687
		R^2	0.0558	0.0205	0.268	0.306
	TS	K_{20}, d^{-1}	--	--	--	--
		θ_{20}	--	--	--	--
		Area, m^2	138	138	138	138
		Depth, m	0.700	0.700	0.700	0.700
		C*, mg/L	--	--	--	--
		k, m/d	8.22	8.22	8.22	--
		P	--	--	3.00	--
		k_0, d^{-1}	--	--	--	3.00
		b, d^{-1}	--	--	--	10.0
		Sum of the square	82,420,449	71,452,072	71,452,072	32,695,892
		R^2	0.0994	0.147	0.147	0.114
	TN	K_{20}, d^{-1}	--	--	--	--
		θ_{20}	--	--	--	--
		Area, m^2	334	1,271	156,163	1,232,341
		Depth, m	0.700	0.700	0.700	6,251
		C*, mg/L	--	247	190	--
		k, m/d	0.179	0.681	83.6	--
		P	--	--	0.16	--
		k_0, d^{-1}	--	--	--	2,951
		b, d^{-1}	--	--	--	49,588
		Sum of the square	232,006	57,210	55,959	67,038
		R^2	0.132	0.464	0.0223	0.0365

R^2 values indicate the correlation between the predicted and measured values of effluent concentrations. Initial values were selected based on reported typical values in the literature.

Table 4. 8. Contaminant removal models approach for the FWS-CTW.

FWS-CTW						
	Water quality parameter	Contaminant removal models				
		Model parameters	Plug-flow	Modified k–C*	TIS (P k–C*)	TDR
			Adjusted value	Adjusted value	Adjusted value	Adjusted value
High precipitation period	TS	K_{20}, d^{-1}	--	--	--	--
		θ_{20}	--	--	--	--
		Area, m^2	138	47.7	9.21	72.0
		Depth, m	1.10	1.10	1.10	0.570
		C*, mg/L	--	--	--	--
		k, m/d	0.190	0.950	0.18	--
		P	--	--	1.06	--
		k_0, d^{-1}	--	--	--	2.53
		b, d^{-1}	--	--	--	6.30
		Sum of the square	385,112	260,364	247,584	262,653
		R^2	0.0189	0.00870	$3.25x10^{-5}$	$1.86x10^{-4}$
	NH_4	K_{20}, d^{-1}	0.600	0.540	0.00387	--
		θ_{20}	1.12	9.92	14.4	--
		Area, m^2	381	362	2.45	138
		Depth, m	3.04	2.89	1.10	1.10
		C*, mg/L	--	1.20	1.50	--
		k, m/d	--	--	--	--
		P	--	--	3.00	--
		k_0, d^{-1}	--	--	--	3.00
		b, d^{-1}	--	--	--	10.0
		Sum of the square	47.5	26.4	17.3	448
		R^2	0.00867	0.0118	0.512	0.120
Low precipitation period	TS	K_{20}, d^{-1}	--	--	--	--
		θ_{20}	--	--	--	--
		Area, m^2	138	53.0	138	$1.67x10^{11}$
		Depth, m	1.10	1.10	1.10	1.10
		C*, mg/L	--	--	--	--
		k, m/d	0.210	1.05	2.74	--
		P	--	--	3.00	--
		k_0, d^{-1}	--	--	--	$1.61x10x^8$
		b, d^{-1}	--	--	--	$5.94x10^9$
		Sum of the square	6,198,300	2,584,745	2,843,504	1,636,612
		R^2	0.0628	0.147	0.147	0.00110
	TP	K_{20}, d^{-1}	--	--	--	--
		θ_{20}	--	--	--	--
		Area, m^2	286	1,289	338	49.2
		Depth, m	1.10	1.10	1.10	0.390
		C*, mg/L	--	0.560	0.56	--
		k, m/d	0.0683	0.310	0.0800	--
		P	--	--	7.72	--
		k_0, d^{-1}	--	--	--	2.45
		b, d^{-1}	--	--	--	3.14
		Sum of the square	2.74	0.800	0.800	2.28
		R^2	0.0399	0.0245	0.0794	0.119

Table 4. 8. (cont'd)

R^2 values indicate the correlation between the predicted and measured values of effluent concentrations. Initial values were selected based on reported typical values in the literature.

4.4. Conclusions

The HCTW described herein successfully treated wastewater from the thermophilic anaerobic digester and was resilient to the changes in precipitation that occurred during this study. The HCTW 1) had sufficient water storage capacity even during the high precipitation period and 2) achieved high removal efficiencies that allow the reuse of the treated water. However, due to large variability in the data, loading charts and contaminant removal models were not able to describe a narrow central tendency in the treatment performance of the HCTW. Importantly, these data are a base for accumulating knowledge and expertise in the application of CTWs in Costa Rica, in terms of water storage capacity, treatment performance, and estimation of design parameters.

The HCTW had better performance during the rainy period, likely due to dilution from precipitation. In addition, high evapotranspiration during the dry period likely increased effluent concentration from the wetlands. Recently, Costa Rica has been experiencing unusual increases in rainfall intensity. The amount of water precipitated per year is similar, but that amount of water is falling during shorter periods (September to October versus July to December) according to meteorological records. This aspect is important to consider in the hydrological balance and treatment performance for future CTWs design.

The hybrid wetland system was important for total treatment performance. The VSSF-CTW received high strength liquid digestate and demonstrated high treatment efficiency. The geotextile membrane provided a low-cost, low-maintenance approach to improving the performance of the VSSF-CTW, as the membrane prevented direct contact of these sediments, facilitated the collection of sediment, and reduced potential clogging. Then, the FWS-CTW served as a polishing step for the effluent from the VSSF-CTW. The HCTW-treated water was considered reclaimed as the use of this water (e.g., irrigation or reuse at the SPAD) did not strictly have to meet discharge standards for water bodies in Costa Rica.

Natural systems such as the HCTW evaluated here need longer periods of evaluation (years) for establishing central tendencies in the treatment. The limited time (28 weeks from August 2015 to March 2016), unsteady weather conditions, and dynamic influent concentrations likely restricted performance analysis and parameter estimation. Kadlec (2000) emphasized first-order models are useful for designing, but inadequacies in estimating or selecting parameter values are highly influenced by the variability of unpredictable events such as fluctuation in input flows and concentration, changes in internal storages, weather, animal activity, and other ecosystem factors.

4.5. Acknowledgments

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CHAPTER 5: CLOGGING IN TROPICAL VERTICAL SUBSURFACE FLOW CONSTRUCTED TREATMENT WETLANDS

Abstract: Clogging of media in vertical subsurface flow constructed treatment wetlands (VSSF-CTWs) can compromise both treatment performance and the wetland's useable lifespan. The goal of this study was to evaluate the longevity of sand media in a tropical VSSF-CTW. Wastewater was pretreated through anaerobic digestion and solid-liquid separation prior to entering the wetland. A geotextile membrane, placed at the inlet area of the VSSF-CTW, further reduced solids entering the wetland and mostly prevented direct contact of solids with the sand media. Infiltrated water, collected at the bottom of the wetland, was recirculated through sprinklers over the surface treatment area of the VSSF-CTW. Under this operation, the VSSF-CTW has not presented signs of clogging after three years. In fact, mass balance of the total solids indicated a constant void space of 20 of 30 m^3 in the filter media. Additionally, the role of plant growth in clogging was examined. At the field scale, no differences in volatile solid accumulation, root development, and infiltration rates in the filter media were found based on plant type. However, in laboratory-scale columns, columns planted with *Canna* and *Cyperus* exhibited different infiltration rates, despite similar volatile solid accumulation and root biomass, indicating that root morphology of *Canna* could be favorable to infiltration.

5.1. Introduction

VSSF-CTWs have been commonly utilized for decentralized wastewater treatment throughout Europe and Asia (Kadlec and Wallace 2009, Wu, Kuschk et al. 2014). VSSF-CTWs consist of filter media, usually sand or gravel, planted with emergent plants. The filter media provides a structural base for the emergent plants and a fixed surface upon which microbial communities can grow (Tilley, Luethi et al. 2008). Wastewater flows vertically through the entire height of the filter media prior to discharge (Kadlec and Wallace 2009). Flow in VSSF-CTWs can be either downward or upward, with either intermittent or continuous application of wastewater. In general, VSSF-CTWs in both tropical and subtropical climates are effective at removing organic matter, total suspended solids (TSS), and nutrients. In Egypt, a VSSF-CTW planted with *Canna indica*, *Phragmites australis*, and *Cyperus papyrus*, treated municipal wastewater with removals of 92% of TSS, 88% of biochemical oxygen demand (BOD), 53% of total nitrogen (TN), and 62% of total phosphorus (TP) (Abou-Elela and Hellal 2012). In small communities, VSSF-CTWs have been used as secondary and tertiary treatment facilities for septic tanks (Garcia-Perez, Harrison et al. 2011), up-flow anaerobic sludge bed (UASB) reactors (de la Varga, Díaz et al. 2013), and anaerobic digesters (Comino, Riggio et al. 2013). For example, a VSSF-CTW filled with 0.80 m of granitic gravel (3 - 6 mm), operating in series with a horizontal subsurface flow constructed treatment wetland (HSSF-CTW), was used to treat effluent from a UASB reactor that contained 1,558 ± 1,023 mg COD/L and 129 ± 88 mg TSS/L. Even with a high surface loading rate (18 ± 13 g BOD/m^{2}d), the hybrid system of wetlands achieved effluent concentrations of 448 ± 541 mg COD/L and 17 ± 15 mg TSS/L (de la Varga, Díaz et al. 2013).

In intermittently-fed VSSF-CTWs, physicochemical and biological processes occur mostly under aerobic conditions (Knowles, Dotro et al. 2011). Physical treatment processes, such as sedimentation, entrapment, and adsorption, remove solids, organic matter, and nutrients from wastewater (Garcia, Rousseau et al. 2010). Biological processes that contribute to wastewater treatment include microbial degradation of organic matter and uptake of nutrients by microorganisms and plant roots (Hua, Li et al. 2013). Roots provide habitat, carbon substrates, and oxygen to rhizospheric microbial communities (Cao, Gregson et al. 1998, Garcia, Rousseau et al. 2010). Due to predominantly aerobic conditions present in intermittently-fed, VSSF-CTWs, this type of CTW removes more organic pollutants from wastewater than HSSF-CTWs on a per area basis, as HSSF-CTWs tend to operate under mostly anoxic or anaerobic conditions (Zhang, Jinadasa et al. 2015).

Wastewater treatment by subsurface flow (SSF)-CTWs can be adversely impacted by clogging (Pedescoll, Uggetti et al. 2009), which decreases porosity, limits hydraulic conductivity, and limits oxygen transport through the filter media (Zurita, De Anda et al. 2009, Hua, Zhu et al. 2010, Nivala, Knowles et al. 2012). Preventive strategies to minimize clogging, including modifying operation and design of the wetland, aim to prolong the lifespan of SSF-CTWs, which is commonly estimated to be 10 to 15 years (Wallace and Knight 2006). With time, filtration of solids by the filter media, chemical precipitation, growth of microorganisms, and root growth can increase clogging (Suliman, French et al. 2006). Hua, Li et al (2013) modeled pore space reduction as a function of accumulation of fixed and volatile solids, microbial growth, and emergent plant roots. Accumulation of fixed solids dominated the clogging process, occupying 70 to 80% of the pore space (Hua, Li et al. 2013). Chemical precipitates of phosphorus and metals can also occupy void spaces in the filter media and thereby also contribute to clogging (Suliman, French et al. 2006). Processes contributing to clogging by organic matter (i.e., volatile solids) are dynamic and complex, as volatile solids are degraded and generated by microbial growth and death (Leverenz, Tchobanoglous et al. 2009). Intermittent application of wastewater in SSF-CTWs can promote both growth of microbes and decay of microbial biomass, thus organic matter and microbial growth are generally considered minor for long-term clogging (Kadlec and Wallace 2009, Hua, Li et al. 2013). Emergent plant roots are considered by some to contribute to clogging, as roots occupy pore spaces in the filter media (Pedescoll, Corzo et al. 2011, De Paoli and Sperling 2013). Pedescoll, Corzo et al. (2011) concluded that *Phragmites australis* root contributed to clogging, as hydraulic conductivity and porosity were lower by 35% and 10%, respectively, in planted horizontal subsurface-flow wetlands than in unplanted wetlands. However, Hua, Zhao et al. (2014) concluded that roots only contributed to clogging during the first stage of plant growth; subsequently, root growth opened new pore spaces in the filter media. Therefore, the debate regarding the role of root growth on clogging is ongoing, as root growth has been reported to open clogged filter media (Wang, Xu et al. 2008) and create preferential pathways for infiltration (Torrens, Molle et al. 2009) or fill pore voids and reduce hydraulic conductivity (Pedescoll, Corzo et al. 2011, De Paoli and Sperling 2013). However, previous studies do not classify roots based on their morphology or compare how different morphologies of roots contribute to clogging. Most studies evaluated how porosity and hydraulic conductivity, as indicators of clogging, differ between planted and non-planted wetlands.

Clogging can be modeled as a function of solids, microbial growth, and root growth. Such models have indicated that reduction of pore spaces in filter media was mostly due to inorganic (or fixed) solids, which occupied 70 to 80% of the pore space, while microbial growth and emergent root plants only represented 6 to 8% of clogged pore space (Hua, Li et al. 2013). Other studies have also found that microbial communities and roots played only a minor role in clogging (Langergraber, Haberl et al. 2003, Sani, Scholz et al. 2013). For this reason, some models only consider inorganic solids; however, understimation of solids accumulation can occur due to incomplete degradation of organic solids (Sani, Scholz et al. 2013). Validation of these models with full-scale VSSF-CTWs has been limited due to difficulty in quantifying parameters without altering the wetland filter in full-scale systems. For example, microbial growth and density, root growth and density, and other microbial-root dependant patameters are typically assumed (Sani, Scholz et al. 2013). The time to clogging model (t_c model) considers void space in the filter media (V_o, cm^3), bulk density of accumulated fixed and volatile solids ($\rho_{b,solids,}$, mg/cm^3), moisture content of the accumulated solids ($MC_{wet\ basis}$, %), influent flow rate (Q, L/d), and the inlet and outlet TSS concentration (C_i and C_e, mg/L), as shown in equation 5.1. While this model simplifies clogging to a solids mass balance and ignores biological or chemical factors (Nivala, Knowles et al. 2012), this model has ben repeatedly used to approximate clogging due to filtration (Langergraber, Haberl et al. (2003), Zhao, Sun et al. (2004), Kadlec and Wallace (2009), and Hua, Zhu et al. (2010)). However, a k factor accounting for the degradation of volatile solids can be used for adjusting the void space in the filter media before estimating the time to clogging (Zhao, Sun et al. 2004).

$$t_c = V_o \times \rho_s \times \frac{(1-MC)}{Q \times (C_i - C_e)} \qquad [5.1]$$

The VSSF-CTW described herein received liquid digestate from an anaerobic digester. The liquid digestate was classified as high strength wastewater due to its high organic content (24.7 – 37.2 g COD/m^2/d) and suspended solid content (16.5 – 24.8 g TS/m^2/d). These loads were higher than recommended for temperate wetlands (e.g., 20 g COD/m^2d and 5 g TSS /m^2d) (Winter and Goetz 2003, Kadlec and Wallace 2009). The VSSF-CTW, planted with *Cyperus papyrus*, *Canna indica*, and *Iris graminea*, removed 87% of TS, 99% of chemical oxygen demand (COD), 97% of TN, and 99% of TP (Aguilar Alvarez, Bustamante Roman et al. 2016). The goal of this study was to evaluate longevity of sand media in the tropical VSSF-CTW by measuring solid accumulation, root development, and infiltration. Additionally, a laboratory-scale VSSF-CTW study was conducted to complement the field study, with the intent of identifying how different root structures affect clogging.

5.2. Material and Methods

A solar-powered anaerobic digester and hybrid constructed treatment wetland was built at the Fabio Baudrit Experimental Station, in Alajuela, Costa Rica (10.00 m N, -84.26 m W) in 2013 (Aguilar Alvarez, Bustamante Roman et al. 2016). The hybrid constructed treatment wetland consisted of a VSSF-CTW followed by a free water surface wetland (FWS-CTW). This paper focuses on the VSSF-CTW, which received liquid digestate from the digester starting in March 2013. This study was conducted from August 2015 to March 2016.

5.2.1. System description

The VSSF-CTW has an area of 12.0 x 12.0 m on the surface and 9.00 x 9.00 m on the bottom (with a median area of 138 m^2). The media is 1.10 m deep. Clay was used as a liner. The media profile from the bottom to the surface is 0.20 m of 12 – 20 mm stone, 0.20 m of 4 – 8 mm pea gravel, and 0.70 m of 0.75 – 2.0 mm coarse sand (32% porosity). From Monday to Friday, one cubic meter of liquid digestate was discharged daily into the VSSF-CTW through an inlet pipe (diameter = 0.05 m) located at the North side of the cell. Liquid digestate discharged onto a 2.0 x 2.0 m geotextile membrane (GT 131, Skaps, Athens, GA), which was installed on the surface of the treatment media. The geotextile membrane was surrounded by cement curbs (height = 0.25 m) that allowed 1 m^3 of wastewater to pool (Figure 5.1). The geotextile membrane had an apparent opening size of 0.30 mm, a rate of 0.102 $m^3/s/m^2$, and a permittivity of 2.20 s^{-1}. After the geotextile membrane, wastewater passed through the filter media by gravity. The wastewater was collected by distributed grid of drainage pipe (0.075 m) in the bottom layer of the cell and conveyed to a well pump (model WS V52 from Franklin Electric, Fort Wayne, IN). The collected, partially-treated wastewater was recirculated at a flow rate of 110 L/min from 2:00 am to 4:00 am daily. The recirculation ratio, as the volume recirculated (13.2 m^3) to the volume of water in the cell (6.15 to 37.4 m^3), varied from 0.46 to 2.8. Recirculation was scheduled during the night to reduce loss of water due to evapotranspiration. Recirculated wastewater was sprayed from four sprinklers located 1.60 m above the ground and at each corner of a square (7.00 × 7.00 m) centered in the wetland. Daily, the pump transferred one cubic meter of treated wastewater into the free water surface constructed treatment wetland.

In 2012, the VSSF-CTW was planted with a grid of randomly-assigned 1 m^2-blocks, with 10 blocks per plant species and 10 unplanted control plots. Plant types were chosen based on availability and previous use in tropical treatment wetlands; initial plantings were *Juncus effuses* (common rush), *Coix lacryma jobi* (Job's tears), *Cyperus papyrus* (papyrus), *Iris graminea* (dwarf iris or grass-leaved iris), and *Canna indica* (wild canna lily or African arrowroot). After three years, at the commencement of this study, *Cyperus papyrus*, *Iris graminea*, and *Canna indica* were thriving in the wetland; the other three species had not survived (Figure 5.1). Excluding the two-meter strip in which the geotextile was installed, the VSSF-CTW was evenly divided in three zones, each three meters wide, from North to South (Figure 5.2). In each zone, one plot for each plant and one control plot was selected for evaluating solid accumulation, root development, and infiltration. In each planted and control plot, infiltration tests were performed at multiple positions, at 0.0 m, 0.5 m, and 1.0 m from the center of the plot.

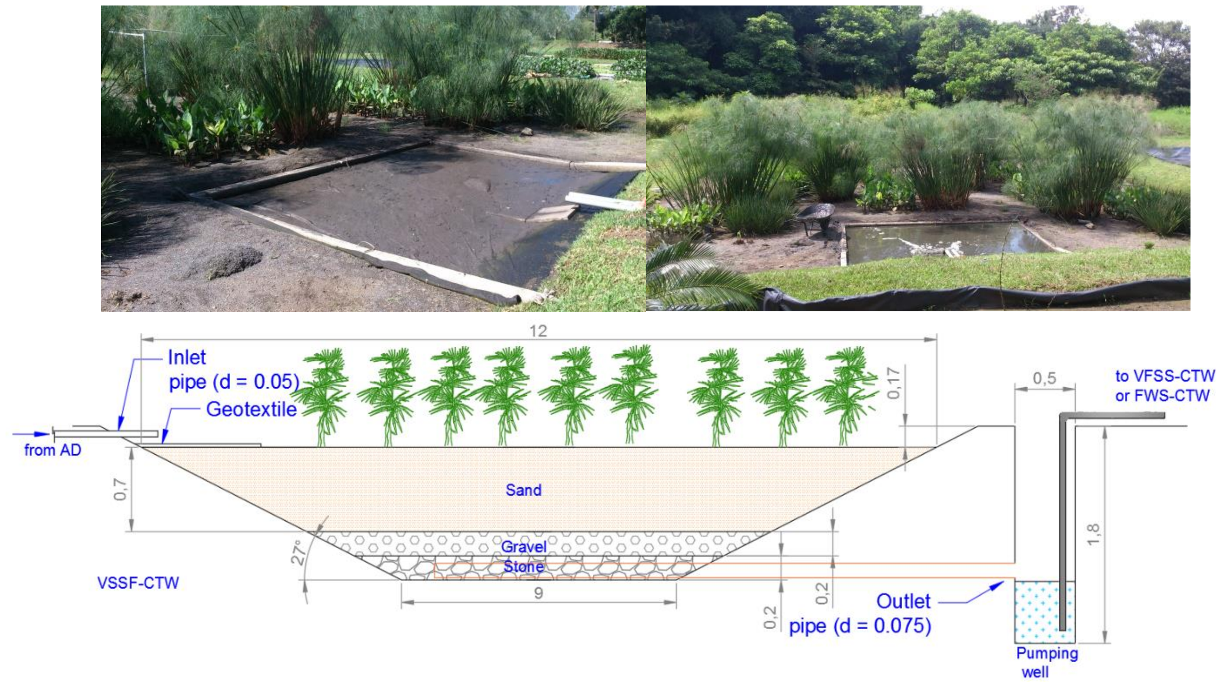


Figure 5. 1. Full-scale VSSF-CTW: Photos of the location of the inlet and geotextile membrane and the health of the plants during this study. Schematic of the full-scale vertical-flow wetland (all lengths are in meters). Pictures were taken in August 2015.

Figure 5. 2. Division of the VSSF-CTW per zones. Picture was taken in March 2013, when the VSSF-CTW was recently built.

5.2.2. Laboratory-scale VSSF-CTW study

Twenty four columns of 0.30 m depth and 0.28 m diameter were filled with coarse sand from the full-scale VSSF-CTW (Figure 5.3). The column depth was chosen based on studies reporting clogging to normally occur within 0.20 m of the surface (Hua, Zhu et al. 2010). Columns were planted with *Canna*, *Cyperus*, and *Iris* from the VSSF-CTW. Triplicate columns were planted with the following treatments: only *Canna*, only *Cyperus*, and only *Iris*; *Canna and Cyperus*, *Canna and Iris, Iris and Cyperus, Canna and Iris and Cyperus,* and control columns without plants. Daily, from Monday to Friday, liquid digestate was applied at a rate of 230 mL/d, the same loading as the full-scale system on a per area basis (1 m^3/d). A PVC outlet with valve (diameter = 0.0127 m) was installed at the bottom of each column. Columns were drained daily by opening the outlet. Columns were constructed and planted in June 2015. Plants were allowed to grow for two months prior to measurements; during this period plants were irrigated with tap water. Nutrients in the filter media were sufficient for plant growth, as indicated by establishment of plants in all columns. This study was conducted in a non-climate-controlled room and room´s temperature depended on environmental conditions; room temperatures ranged from 15.0 to 33.0°C. Ambient sunlight from the open sides of the room was supplemented with fluorescent lights. From 6:00 am to 5:00 pm, supplemental light was provided using fluorescent lights hung 1.30 m above the table (Figure 5.3).

Figure 5. 3. Laboratory-scale VSSF-CTW. Picture was taken in July 2015, during establishment of plants in the columns.

5.2.3. Solids accumulation in the filter media

Media core samples were collected from both the field-scale and column-scale wetlands. For the column experiment, to avoid alteration of the filter media during the experiment, core samples were collected only at the conclusion of the experiment (February 2016). For the VSSF-CTW, core samples were collected in November 2015 and January, February, and March 2016. Sand cores were taken from three positions (0.0 m, 0.5 m, and 1.0 m from the center) in treatment blocks of the field-scale wetland. Sand cores were collected by inserting a 0.025-m diameter soil corer to a depth of 0.30 m. Samples were collected in plastic bags and were kept at 4°C until analysis. Cores were divided into multiple layers: the surface to 0.050 m deep, 0.050 to 0.10 m deep, 0.10 to 0.15 m deep, and 0.15 to 0.30 m deep. Roots were removed from each sand layer. Each sand layer was homogenized and a sample from each layer was analyzed for total, fixed, and volatile solids (Hach method #8276). Sand samples were weighed and put in aluminum dishes (VWR®, catalog number 25433-008) and oven dried in a StabilTherm gravity oven (model OV-12A from Blue M, East Troy, WI) at 100°C for 24 hours to determine total solids. Sand samples then were put in a StableTemp furnace (model CBFS516A from Cole-Parmer, Vernon Hills, IL) at 500°C for 30 minutes to determine volatile solids and fixed solids. Consequently, the measurements for fixed and total solids in the core samples included the sand media.

A mass balance of fixed solids was conducted to estimate void space in the filter media of the VSSF-CTW and the time before clogging using the approach by Langergraber, Haberl et al. (2003), Zhao, Sun et al. (2004), and Hua, Zhu et al. (2010). The initial void space in the filter media (V_o, m^3) was estimated by multiplying the porosity of coarse sand (Ɛ), the depth of filter media (h), and the median surface area (A) (equation 5.2). Influent flow rate (Q, L/d), inlet and outlet total solids concentration (C_i and C_e, mg/L), bulk density of accumulating solids (ρ_s, mg/cm^3), moisture content of the accumulating solids (MC, %), and bed operating time (t, d), were used to estimate the wet volume of captured solids (equation 5.3). Total solids in the influent and effluent were measured from weekly samples taken at the VSSF-CTW (Hach method #8276). The available void space in the filter media (V_t, m^3) immediately after wastewater application was estimated as the difference between the intial void space and the wet volume of captured fixed and volatile solids. Available void space in the filter media after resting (V_{01}, m^3) assumed that organic solids (e.g., volatile solids) were completely degraded and was calculated by determinng k (%) (equation 5.4) by measuring the ratio of volatile:total solids in the influent. Table 5.1 summarizes the parameters used in this study. The time to clogging was estimated with equation 5.1.

$$V_o = \varepsilon \times h \times A \qquad [5.2]$$

$$V_{t,s} = \frac{Q \times (C_i - C_e)}{\rho(1 - MC)} \times t \qquad [5.3]$$

$$V_{01} = V_0 - V_{t,s}(1-k) \qquad [5.4]$$

Table 5. 1. Given parameters for estimating of the void space in the filter media and the time before clogging at the full-scale VSSF-CTW.

Parameter	Unit	Values	Source
Porosity of coarse sand, Ɛ	%	32	Measured
Depth of filter media, h	M	0.70	Measured
Median surface area, A	m^2	138	Measured
Flow rate, Q	L/d	660 – 1,100	Measured
Inlet TS concentration, Ci	mg/L	2,101 – 8,155	Measured
Outlet TS concentration, Ce	mg/L	255 – 3,770	Measured
Density of solids, ρ_s	mg/cm^3	1050	(Zhao, Sun et al. 2004)
Moisture content of SS, MC	%	98.8 – 99.7	Measured
Bed operating time	d	7.0	Measured
Organic substrate of solids, k	%	8.48 – 55.6	Measured

5.2.4. Root development in the filter media

Root biomass was collected to evaluate whether infiltration changed due to presence of roots. Roots were collected from core media samples. Roots were carefully removed from the sand, placed on a sieve with 0.05-cm openings, and shaken and rinsed until sand was visibly removed from the roots. Root samples were oven dried in a StabilTherm gravity oven (model OV-12A from Blue M, East Troy, WI) at 100°C for 24 hours to determine dry mass.

5.2.5. Infiltration in the filter media

Modified infiltration tests, similar to those by (De Paoli and Sperling 2013), were performed to evaluate the extent of clogging in the wetland and to compare infiltration rates based on plant treatment. For the column experiments, infiltration tests were performed in August, September, and November 2015 and February 2016. First, columns were saturated with tap water. Once saturated, 0.032 m of tap water was applied to each column. Next, the outlet was opened and the time for infiltrating 0.032 m of tap water was recorded. Three repetitions per column were conducted. The column experiment was conducted to determine differences between the plant treatments when digestate was directly applied to the treatment media (i.e., without geotextile separation of solids). For the VSSF-CTW, infiltration tests were conducted in January, February, and March 2016. The VSSF-CTW was not saturated prior to testing; instead, infiltration tests were performed under ambient moisture content in the media. In each zone, infiltration was conducted for each plant treatment and control plot. In each selected plot, three infiltration tests were performed at multiple positions (0.0 m, 0.5 m, and 1.0 m from the center of the plot). The double ring infiltration method was modified due to space constraints in the VSSF-CTW. A 0.30-m long PVC pipe (diameter = 0.10 m) was inserted 0.20-m deep in the sand. Prior to the test, tap water was poured inside the PVC pipe until a constant infiltration rate was achieved. The time for infiltrating the 0.25 m of tap water at a constant rate was recorded as an estimate of the infiltration rate. The experiment was conducted to determine differences based on time, zone (representing distance from the inlet), plant species, and position (distance from the center of the block).

5.2.6. Treatment performance of the VSSF-CTW

For the column experiments, COD (Hach method #8000) and TS (Hach method #8276) were measured from the inlet and outlet of the columns. One measurement per month was conducted in August, September, November 2012, and February 2016, before performing the infiltration test. Samples were collected in 50 mL plastic containers, capped with a lid, and stored at 4°C until analyses. A DRB 200 reactor (Hach product #LTV082.53.40001) and a DR 900 multiparameter handheld colorimeter (Hach product #9385100) were used to digest and quantify

COD concentration (low range digestion vials from 0 to 150 mg COD/L, Hach product #2565025). Treatment performance was compared amongst plant treatments.

5.2.7. Statistical analysis

Statistical analysis of data was conducted using JMP®, Version 10.0.0 (SAS Institute Inc., Cary, NC). Normality of the data was evaluated using the Shapiro-Wilk test. Parametric tests were conducted in case of normality; otherwise, non-parametric tests were conducted (Table 5.2). All tests were evaluated at an alpha level of 0.05. In addition, multiple linear regressions were conducted for estimating correlations between infiltration rates, volatile solids, and root biomass.

Table 5. 2. Statistical analysis.

Experiment	Scale	Type of data	Statistical test	Goal
Solids accumulation in the filter media	Column	Non-normal	Kruskal-Wallis	To determine differences of solids accumulation at different depths and plant treatment.
	Field	Non-normal	Friedman´s test Kruskal-Wallis	To determine differences of solids accumulation based on time. To determine differences among the zone, position, and plant species.
Root development in the filter media	Column	Normal	Turkey-Kramer HSD	To determine differences in root development based on treatment.
	Field	Non-normal	Kruskal-Wallis	To determine differences among zone, root development and plant species.
Infiltration in the filter media	Column	Non-normal	Kruskal-Wallis Friedman´s test	To determine differences based on plant treatments. To determine infiltration changes based on time.
	Field	Normal	Repeated measurements analysis of variance Turkey-Kramer HSD Multiple linear regression	To determine differences based on time. To determine differences based on zone (representing distance from the inlet), plant treatments, and position (distance from the center of the block). To determine correlations between infiltration rates and root biomass, and infiltration as function of root biomass and volatile solids.

5.3. Results and discussion

5.3.1. Full-scale VSSF-CTW

Multiple solid-liquid separation units were used to remove solids from the digestate to minimize solid loading on the VSSF-CTW, including screen separation of solids and liquids, storage, and filtration. After anaerobic digestion, a rotary liquid/solid separation unit (ICAFE ®, Costa Rica, particle size > 0.5 mm in diameter) was used to separate out large solid particles from the digestate. As a result, 28 kg of digestate solids were removed per day; separated solids were used as a fertilizer after composting (Aguilar Alvarez, Bustamante Roman et al. 2016).

Separated solids mainly consisted of undigested fruit seeds (e.g., watermelon and cantaloupe) and plant fibers (Figure 5.4.a). The liquid digestate was then stored in the effluent storage tank, which served as a settling tank. It was necessary to clean the tank in January 2016 due to accumulation of sediments (not measured) to the height of the outlet pipe (Figure 5.4.b).

a b

Figure 5. 4. Preventive strategies applied for reducing solids in the liquid digestate. a. Rotary liquid/solid separation unit; and b. effluent storage tank. Pictures were taken in January 2016.

Finally, the geotextile membrane removed fine particles from the liquid digestate. The membrane dimension (2 x 2 x 0.25 m^3) had sufficient capacity to hold the one cubic meter discharged per day into the VSSF-CTW. According to specifications, the geotextile membrane retained particles larger than 0.30 mm and allowed a flow rate of 0.408 m^3/s. Thus, in the absence of solids, approximately one cubic meter of water should have passed through the membrane in 2.45 s. However, the liquid digestate pooled for much longer while passing through the geotextile (not measured, approximately one to 15 hours). Continuous accumulation of solids in the geotextile membrane reduced the flow rate. Per week, the membrane removed 4.81% of COD, 27.9% of TS, 13.4% of TN, and 19.5% of TP (Table 5.3), creating a thick layer of sediments (Figure 5.5). The membrane was important because it prevented direct contact of solids with filter media and, subsequently, formation of a crust was not observed in any zone of the treatment area of the VSSF-CTW. Moreover, accumulated solids were easily collected from the membrane for further use as a fertilizer (Figures 5.1 and 5.5).

a b c

Figure 5. 5. The geotextile membrane: a. during liquid digestate influent flow; b. during filtration of pooled digestate; and c. during removal of sediments. Pictures were taken in October 2014.

Table 5. 3. Mass removals for COD, TS, TN, and TP at VSSF-CTW *.

	Mass loading into the geotextile	**Mass removal by the geotextile**	**Mass loading after the geotextile**	**Mass after the VSSF-CTW**	**Mass removal by the VSSF-CTW**	**Removal by the VSSF-CTW****
	kg /m^2/week		kg /m^2/week			(%)
COD	8.92 ± 1.08	0.429 ± 0.0871	8.49 ± 1.05	0.0558 ± 0.0268	8.44 ± 1.04	99.4 ± 0.3
TS	5.77 ± 0.734	1.61 ± 0.328	4.15 ± 0.737	(6.17 ± 2.06) x10^{-2}	4.09 ± 0.727	99.0 ± 0.4
TN	1.39 ± 0.145	0.186 ± 0.0298	1.20 ± 0.136	(2.37 ± 0.631) x10^{-3}	1.20 ± 0.136	99.8 ± 0.1
TP	0.148 ± 0.0269	0.0289 ± 0.00587	0.119 ± 0.0254	(1.72 ± 0.978) x10^{-4}	0.119 ± 0.0254	99.8 ± 0.1

* Modified from Chapter 4
** Includes the geotextile membrane

The expected movement of solids in the VSSF-CTW is shown in Figure 5.6. The liquid digestate was discharged above the geotextile membrane and 27.9% of TS were removed. The movement of the wastewater through the filter media was not measured or modeled; however, the filter media beneath the geotextile received high mass loading and the VSSF-CTW removed 99.0% of TS. A mass balance of the solids was conducted in the portion of the wetland beneath the geotextile membrane (4 m^2 and 0.70 m height of the filter media) to estimate the void space in the filter media and the time before clogging (equations 5.1 – 5.4). The mass balance estimated that void space in the media beneath the geotextile would be occupied in a range of 2.93 to 3.34 days (beneath the geotextile series, Figure 5.7.a), based on the loading of solids after the geotextile membrane. Therefore, in a short period, this portion of the wetland was likely clogged. In fact, at the end of the experiment, liquid digestate pooled for longer times above the geotextile membrane, even after the geotextile membrane was cleaned. Based on this estimation, an extra 40 – 50 m^3 of solids entered beneath the geotextile membrane than what it could hold. These solids could become more compacted and/or also diffuse away from the media beneath the geotextile membrane. Therefore, if there was no compaction, then the volume available in the entire treatment area is less than estimated volume of solids. However, due to changes in the operation of the wetland after this study, it was not possible to analyze this portion of the wetland for solids accumulation or infiltration rate.

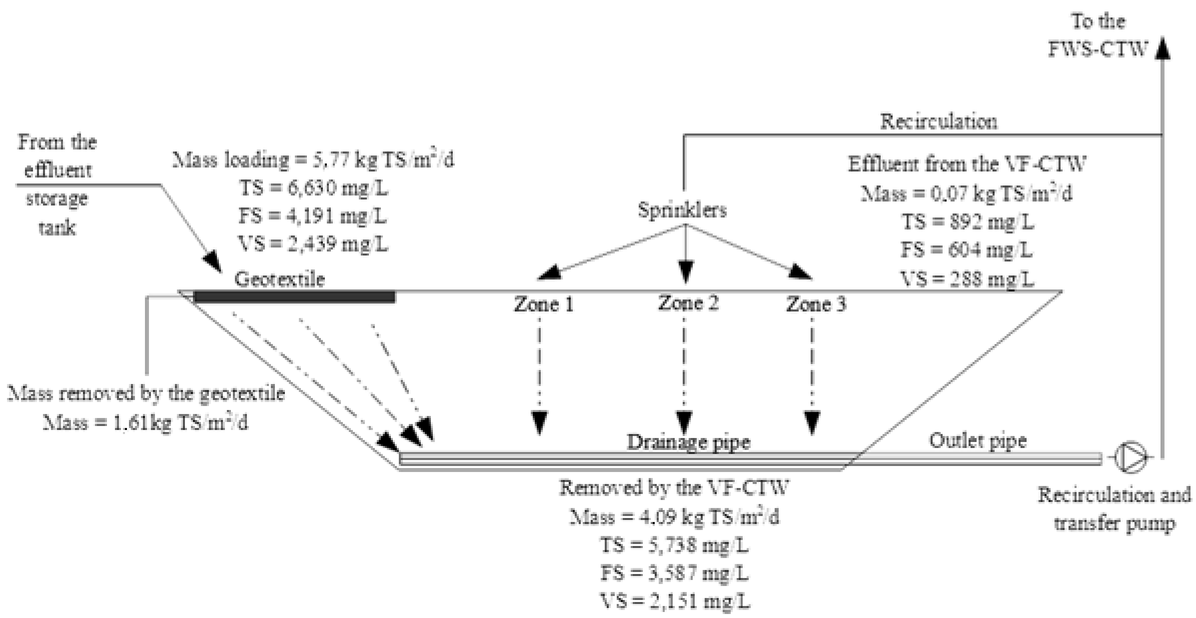


Figure 5. 6. Mass balance on the VSSF-CTW. Figure is not to scale.

The surface of the main treatment area in the VSSF-CTW (zones 1, 2, and 3, Figures 5.2 and 5.6) only received wastewater through recirculation; influent wastewater would have dispersed mostly within the media beneath the geotextile. Consequently, loading of solids in zones 1, 2, and 3 was considerably less than the initial solid loading in the liquid digestate (Figures 5.6 and 5.7.b). A mass balance of the solids was conducted to estimate the void space in the filter media and the time before clogging in zones 1, 2, and 3 (108 m^2 and 0.70 m height of the filter media). The inlet total solids concentration was estimated as the 75th-percentile of the effluent concentration data ($C_i = C_{75e} = 978$ mg/L), while the outlet TS concentrations were set as the effluent concentrations from the VSSF-CTW. Under this condition, void space in the filter media was estimated to be higher than 20 m^3 and relatively constant (main treatment area series,

Figure 5.7.a). This simulation could explain why a crust was never observed in the surface area of the VSSF-CTW. A final estimation of solid accumulation considered the case of an even distribution over the wetland surface area (138 m^2 and 0.70 m height of the filter media) without a geotextile membrane. The VSSF-CTW would then hypothetically become clogged in 80 days (main treatment area if no geotextile were used, Figure 5.7.a).

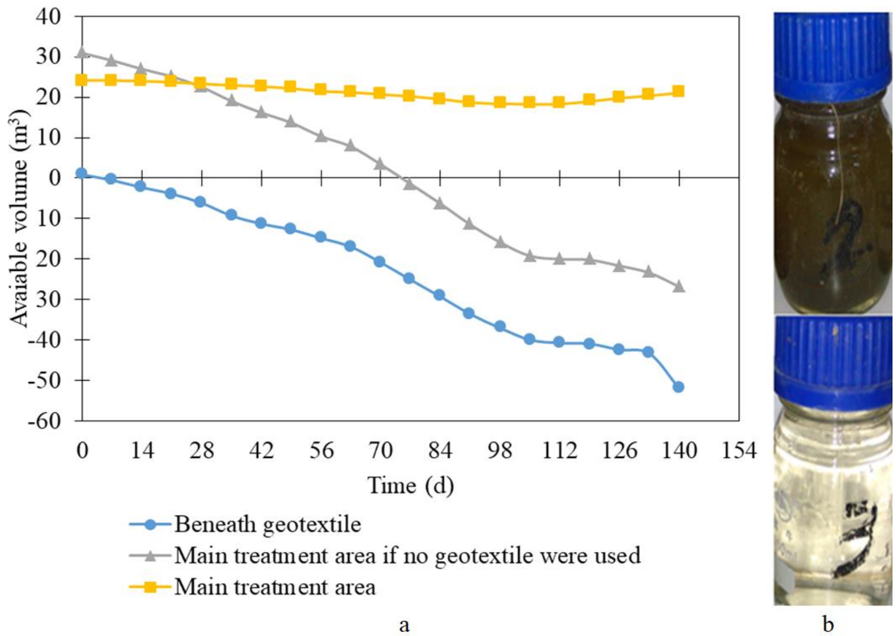


Figure 5. 7. a. Mass balance of the solids to estimate the void space in the filter media and the time before clogging; b. Photos of wastewater before and after the VSSF-CTW.

After three years of operation, the mass balance indicated that fixed solids have not impacted the filter media of the VSSF-CTW. Consequently, a further examination of volatile solids in the media was conducted. In general, volatile solid accumulation for all plant treatments ranged from 4.46 to 12.0 g VS/L (Figure 5.8). Volatile solids concentrations in the media were similar for each plant species and independent of zone and sampling month, as indicated by the p-values >0.05 when the Wilcoxon non-parametric multiple comparisons were conducted (Table

5.4). However, the experimental design was not powerful enough to detect differences in volatile solids accumulation, likely due to limited numbers of replicates. The power of comparisons based on zone and plant treatment ranged from 0.052 to 0.520 (Table 5.4). For example, in zone 1 at February 2016, the comparison of volatile solid accumulation in the media beneath *Iris* plants, as compared to the media beneath *Cyperus* plants, yielded a p-value of 0.0809. However, the power of the statistical comparison for these two plant treatments (0.520) was below the commonly accepted value of 0.8, indicating that it is possible that a difference may have been detected with a greater sample number. Compared with De Paoli and Sperling (2013), our VSSF-CTW was not clogged. In De Paoli and Sperling (2013), a subsurface flow wetland planted with *Typha latofolia* accumulated 35 g VS/L in the area where wastewater was discharged, causing clogging and surface runoff. In addition, De Paoli and Sperling (2013) found 16.8 g VS/L in an unplanted subsurface flow wetland, in which no surface runoff was reported.

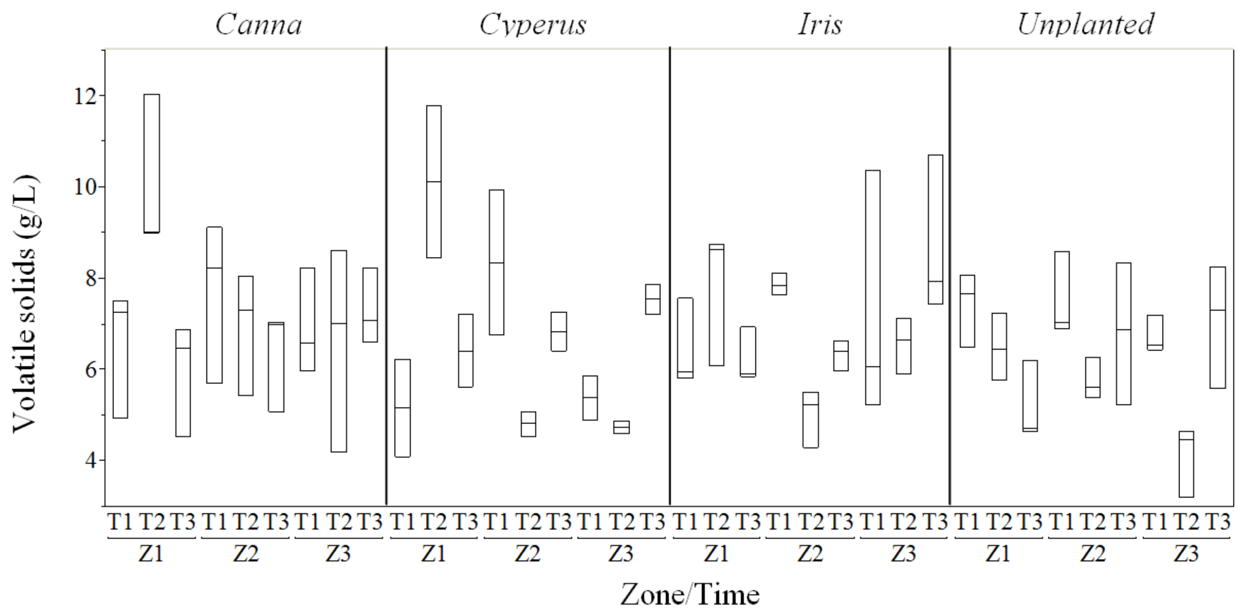


Figure 5. 8. Solid accumulation (as g VS//L) in the VSSF-CTW. T1: January 2016, T2: February 2016, and T3: March 2016. Z1: zone 1, Z2: zone 2, and Z3: zone 3. Statistical analysis applied: Kruskal-Wallis, using Wilcoxon non-parametric multiple comparison at alpha level of 0.05. N = 9, except for *Cyperus* (n=6).

Table 5. 4. Statistical p-value and power for solid accumulation analysis in the vertical-flow wetland.

Plant treatments*	**January 2016**			**February 2016**			**March 2016**		
	Zone 1	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**	**Zone 1**	**Zone 2**	**Zone 3**
p-value	>0.149	>0.662	>0.148	>0.0809	>0.149	>0.0809	>0.382	>0.386	>0.383
Power	0.243	0.0678	0.115	0.520	0.0520	0.421	0.154	0.0699	0.172

Since March 2013, plants have been trimmed to keep them within the predefined one square meter blocks. During this study, *Cyperus* demonstrated the most robust growth. *Cyperus* plants formed a dense culm that did not allow the collection of core samples and roots or conduction of infiltration tests in the center of the blocks without injury to the *Cyperus* plants. In general, *Cyperus* had greater total root biomass than *Canna* and *Iris*, even when a sample from center of the block was not collected for *Cyperus*. *Cyperus* had denser and longer roots, as roots were also found at one meter from the center of the block; in contrast, *Canna* and *Iris* roots were not observed at one meter from the center of the block. In addition, *Cyperus* root biomass was greater at 0.50 m from the center of the block than *Iris* (p=0.0006) and *Canna* (p=0.0013). Finally, *Canna* root biomass was greater than *Iris* root biomass at the center block (p=0.0015) and at 0.50 m from the center of the block (p=0.0134) (Figure 5.9). Similarly, Cheng, Chen et al. (2009) reported higher root growth for *Cyperus* (7,000 g/m^2) than *Canna* (5,500 g/m^2). In addition, Liang, Zhang et al. (2011) demonstrated that *Iris* developed lees root biomass (3.67 g/plant) than *Canna* (7.26 g/ plant).

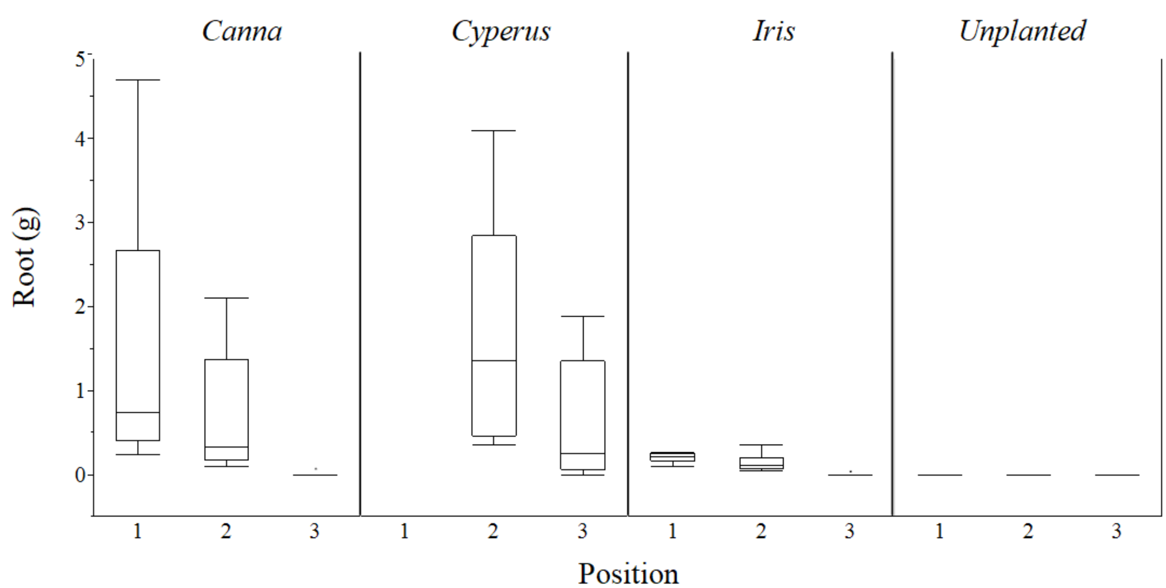


Figure 5. 9. Total dry root biomass per species (n = 9, except for Cyperus (n = 6)) established at the VSSF-CTW. Abbreviations: Iris (IG), Canna (CI), Cyperus (CP), and control (C). Position 1 (center of the block); position 2 (0.50 m from the center of the block); and position 3 (1.0 m from the center of the block).

Infiltration rates for all plant treatments in any zone and among zones with plant treatment were not significantly different, as indicated by the p-values >0.05 when the two way analysis of variance was conducted at alpha level of 0.05 (Table 5.5). In addition, no differences were found based on time (Figure 5.10). This result was expected, as the volatile solids accumulation over zones 1, 2, and 3 was similar. Any difference in volatile solids accumulation in the filter media can impact infiltration rate. For example, De Paoli and Sperling (2013) reported higher hydraulic conductivity (107 m/d) in treatment areas where the VS concentration was 1 g/L, in contrast to 11 m/d in treatment areas where the VS concentration was 35 g/L.

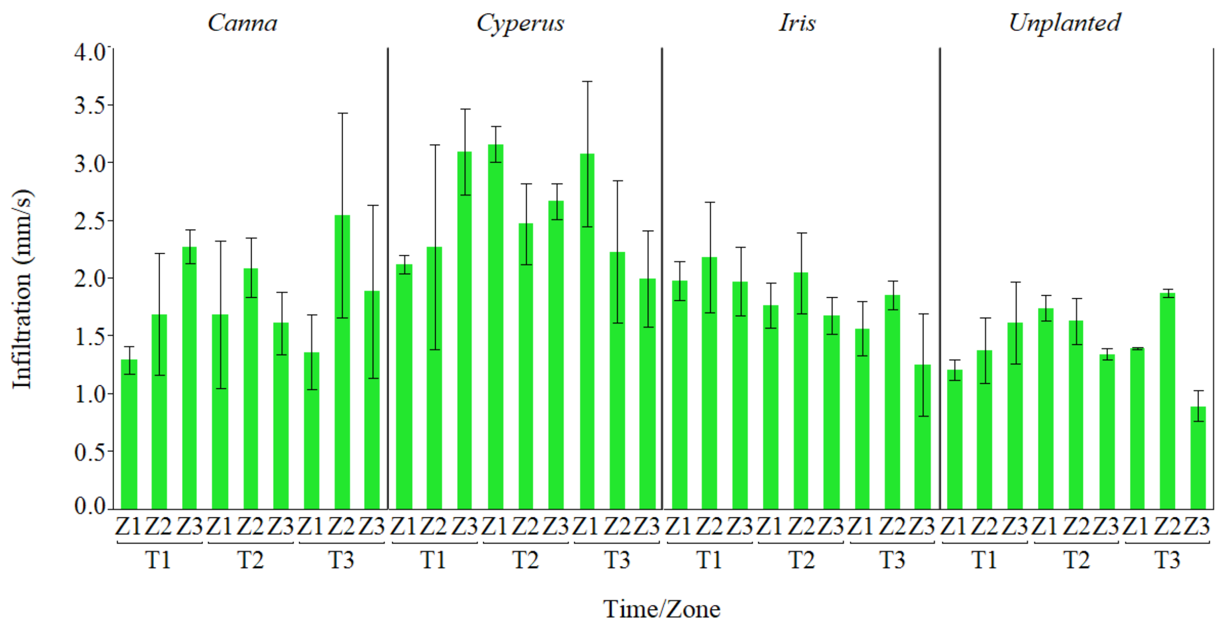


Figure 5. 10. Infiltration rates at the VSSF-CTW (n = 9, except for *Cyperus* (n = 6)). T1: January 2016, T2: February 2016, and T3: March 2016. Z1: zone 1, Z2: zone 2, and Z3: zone 3. Statistical analysis applied: two way analysis of variance at alpha level of 0.05.

Table 5. 5. Statistical p-value and power for infiltration rates in the VSSF-CTW.

Plant treatments*	**January 2016**	**February 2016**	**March 2016**
Zone, p-value	0.084	0.414	0.198
Zone and treatment, p-value	0.734	0.718	0.514

In general, low correlations were found when infiltration was analyzed as a function of volatile solids and root biomass. Considering all plant treatments, volatile solids, and root biomass, the R^2 yielded a low value of 0.25. In fact, low correlations for *Canna* (R^2= 0.24), *Cyperus* (R^2= 0.13), *Iris* (R^2= 0.16), and unplanted blocks (R^2= 0.005) were found. For all cases, power ranged from 0.052 to 0.75, below the desired power of 0.80, and it was less likely to detect higher correlations due to limited numbers of replicates.

Infiltration was dependent on proximity to the main stalk for only *Canna*. It was noticed that infiltration rate increased from 1.0 m from the center of the block to the center of the block for *Canna* ($p = 0.0040$), even though more roots were observed at the center of the plot (Figure 5.9). A similar trend was found for *Cyperus,* as infiltration at 0.50 m from the center of the block was higher than infiltration at 1.0 m from the center of the block ($p = 0.0172$), even though there was higher root biomass at 0.50 m from the center of the block (Figure 5.9). In fact, infiltration rates increased as root biomass increased for *Canna* and *Cyperus*; however, the correlations between root biomass and infiltration rates were low for *Canna* (R^2=0.24) and *Cyperus* (R^2= 0.24) (Figure 5.12). In contrast, *Iris* had similar infiltration rates at the center of the block and at 0.50 m from the center of the block; however, infiltration at 0.50 m from the center of the block was significantly greater than infiltration rate at 1.0 m from the center of the block ($p = 0.0162$), where no roots were found (Figure 5.9). In fact, the correlation between root biomass and infiltration rates was low for *Iris* (R^2= 0.15), and slow and rapid infiltration rates were obtained with a similar root biomass (Figure 5.12). In general, infiltrations rates increased as root biomass increased; however the correlation was low (R^2= 0.25). Despite these low correlations between infiltration rates and root biomass, it was likely that roots had impacted infiltration rates. For example unplanted plots had similar infiltration rates based on position, likely due to uniform similar VS concentration in the filter media (Figure 5.8) and lack of roots occupying pore spaces in the filter media (Figure 5.11).

Figure 5. 11. Infiltration rates per positions at the VSSF-CTW. Position 1 (center of the plant); position 2 (0.50 m from the center of the plant); and position 3 (1.0 m from the center of the plant). Letters A, B, and C indicate significant differences between infiltrations per position per each plant treatment. Statistical analysis applied: Turkey-Kramer HSD at alpha level of 0.05.

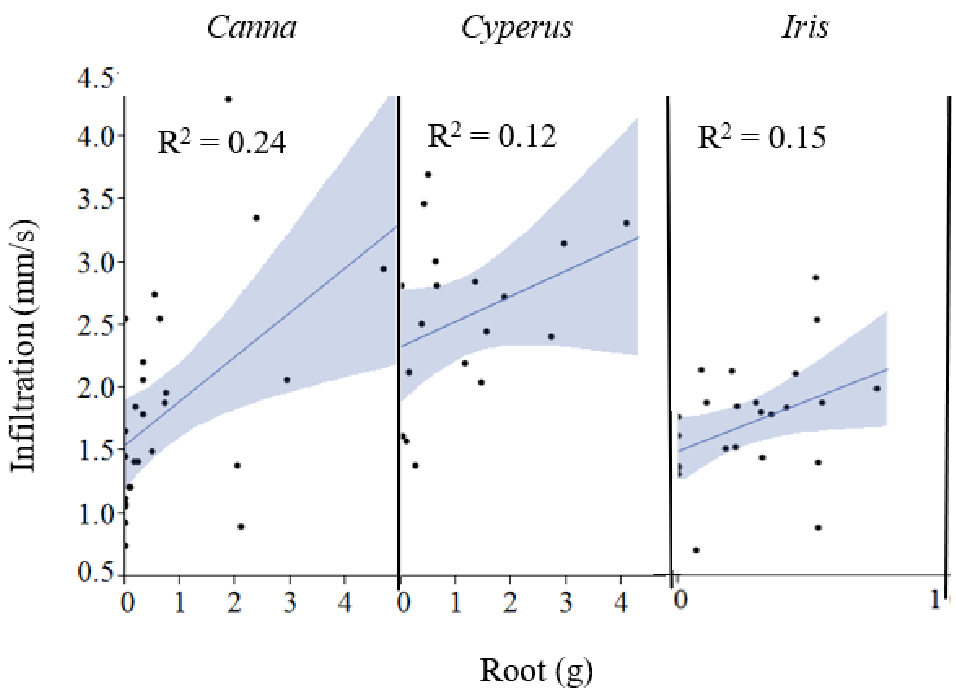


Figure 5. 12. Infiltration rates in the VSSF-CTW as a function of root biomass for *Canna*, *Cyperus*, and *Iris*.

5.3.2. Laboratory-scale VSSF-CTW

The laboratory-scale VSSF-CTWs were built to determine how different root structures affect clogging. This study was limited to only one experimental replicate of columns due to time and budget constraints. Thus, to improve representativeness of columns for estimating clogging would be needed more sets of columns keeping some factors constant (e.g.: column Depth, OLR, plant density) and varying other factors (e.g.: diameter of columns).

The filter media directly received liquid digestate and recirculation was not performed. Consequently, clogging in the laboratory-scale wetlands was affected by a combination of fixed solids, volatile solids, and root growth. In general, COD removal was higher than 98%; however, effluent COD concentrations increased toward the end of this experiment. Similarly, TS removal decreased with time. After accounting for time-dependence, treatment performance did not differ amongst plant treatments for COD ($p>0.247$), for TS ($p>0.286$), and for FS ($p>0.243$). Generally, treatment performance tends to increase during initial accumulation of solids on the surface; however, filtered and trapped solids will eventually limit oxygen transport and reduce the ability of the system to treat the wastewater, especially if a crust is formed (Hua, Zhu et al. 2010). Our data indicates that crust was formed after 30 days, and treatment performance decreased in November 2015. In fact, COD effluent was similar in September and October 2015, however, effluent significantly increased to 54 mg COD/L in November 2015 ($p = 0.0041$, with respect to October 2015), and significantly increased to 140 mg COD/L in February 2016 ($p < 0.0001$, with respect to November 2015). Similarly, TS and FS effluent significantly increased to 4,323 mg TS/L and 1,592 mg FS/L in November 2015 ($p<0.0187$, with respect to October 2015). In our case, it was likely that treatment declined due to the accumulation of solids (Table 5.6).

Table 5. 6. Effluent water quality parameters in the columns (mean ± S.D., n = 24).

Parameter	**Sept-2016**	**Oct-2015**	**Nov-2015**	**Feb-2016**
COD in (mg/L)	4,090 ± 221	5,896 ± 637	11,313 ± 889	7,420 ± 400
COD out (mg/L)	39 ± 14 [a]	40 ± 11 [a]	54 ± 18 [b]	140 ± 19 [c]
COD removal (%)	99.0 ± 0.4	99.3 ± 0.4	99.5 ± 0.4	98.1 ± 0.2
TS in (mg/L)	3,371 ± 154	4,228 ± 581	6,254 ± 335	4,251 ± 1,034
TS out (mg/L)	n.d.	1,861 ± 517 [a]	4,323 ± 1,100 [b]	5,000 ± 1,400 [b]
TS removal (%)	n.d.	55.9 ± 0.2	30.8 ± 0.1	(17.6 ± 0.1)
FS in (mg/L)	1,936 ± 62	2,732 ± 210	4,480 ± 189	3,089 ± 623
FS out (mg/L)	n.d.	1,071 ± 74 [a]	1,592 ± 60 [b]	2,380 ± 169 [c]
FS removal (%)	n.d.	60.8 ± 0.1	64.4 ± 0.1	22.9 ± 0.1

Letters indicate statistical significant mean differences of COD, TS, and FS concentration base on time at $\alpha = 0.05$. No data (n.d.)

The filter media removed from 20 to 65% of fixed solids and 0 to 47% of the volatile solids. In general, concentrations of volatile solids in the top 0.05 m of media were approximately 50% greater than volatile solid concentrations in the lower three layers; however, this difference was only statistically significant for columns planted with only *Iris* (Figure 5.13).

For *Iris*, solids accumulation in the surface layer was significantly different with respect to deeper layers ($0.0131 < p < 0.0306$). It is possible that the experimental design, given the limitations on replicate number, was not powerful enough to detect differences in volatile solids accumulations between media depths for other columns. The power of comparisons based on depth and plant treatment ranged from 0.0950 to 0.358. For example, the comparison of volatile solid accumulation in the surface layer of *Cyperus*-planted columns, as compared to the second layer, yielded a p-value of 0.0656. However, the power of the statistical comparison for these two layers (0.215) was below the commonly accepted value of 0.8, indicating that it is possible that a difference may have been detected with a greater sample number. Higher accumulation in the top layer with respect to deeper layers was expected due to filtration and entrapment of solids by the media (Hua, Zhu et al. 2010). In fact, a crust formation of solids was continuously observed each time the columns were irrigated with liquid digestate. Similar trends were reported by Hua, Zhu et al. (2010), when they observed in a laboratory-scale vertical-flow wetland (0.15 m diameter and 0.40 m depth), that the filter gradually clogged due to the application of 600 mg total solids/L of wastewater. Overall, more volatile solids were accumulated in columns planted with only *Iris* than in the control or other planted columns. For example, concentrations of volatile solids in the top layer of columns planted with only *Iris* ranged from 12.9 and 22.7 g/L; in contrast, accumulation of volatile solids in the top layer of other treatments was significantly lower at 9.98 to 13.7 g/L.

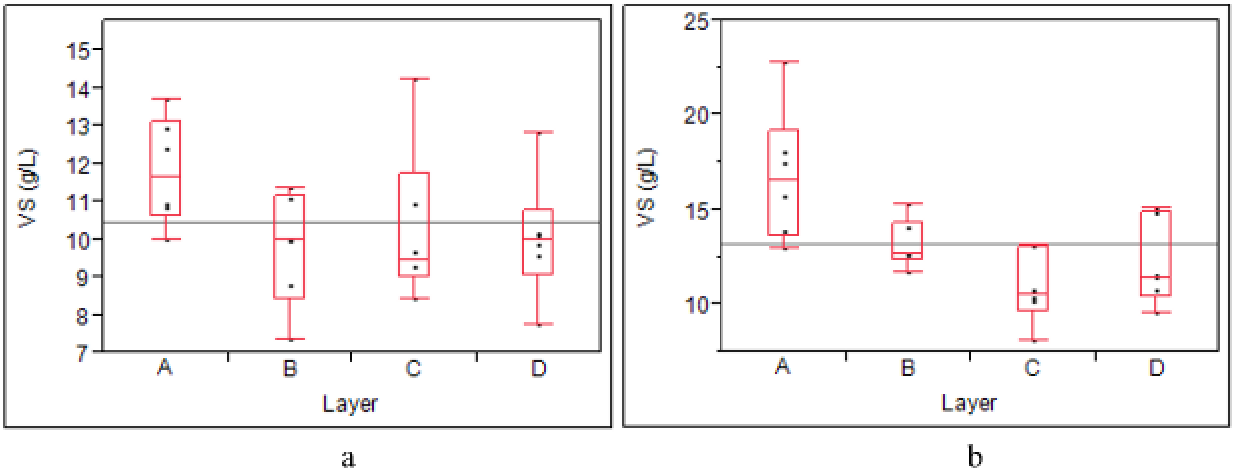


Figure 5. 13. a. VS accumulation unplanted columns and six of the seven plant treatments (*Cyperus*, *Canna-Cyperus*, *Canna*, *Canna-Iris-Cyperus*, *Canna-Iris*, and *Iris-Cyperus*); b. VS accumulation for *Iris* plant treatment. Note. A: from surface to 0.05 m deep, B: from 0.05 to 0.10 m deep, C: from 0.10 to 0.15 m deep, and D: from 0.15 to 0.30 m deep.

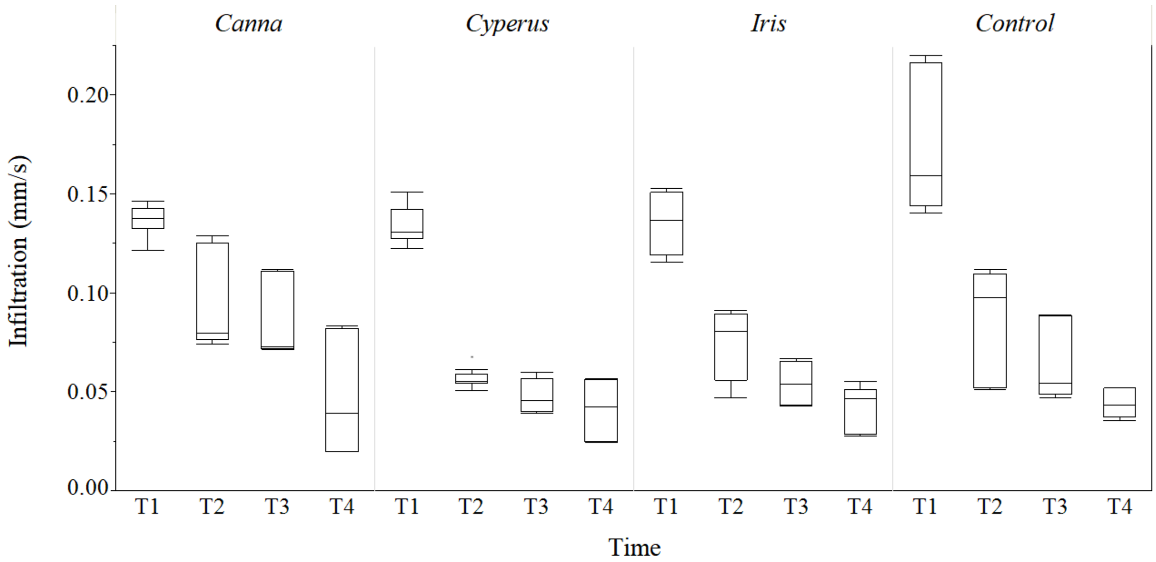
Root development was generally less for individual plant treatments (e.g., *Iris*, *Canna*, *Cyperus*) than for multiple plant treatments (e.g., *Canna-Iris-Cyperus*). Based on comparisons for all pairs using Turkey-Kramer HSD, root development per treatment was ranked as low, medium, or high (Table 5.7). With the exception of the *Canna-Iris-Cyperus* and *Canna-Iris* treatments, all planted treatments were classified as “low” root biomass. The greatest root biomass occurred in columns planted with all three plant species (*Canna-Iris-Cyperus*) and *Canna-Iris*. Importantly, only one plant was planted per species in the column. For example, *Canna* only had one plant, whereas *Canna-Iris-Cyperus* had three plants. In individual species treatments, *Cyperus* developed slightly more root biomass than *Canna* and *Iris*; similar to results obtained by Liang, Zhang et al. (2011), who reported that *Iris* plants grew 3.67 g of roots per plant, as compared to 7.26 g/*Canna* plant and 1.17 g/*Cyperus* plant. Likewise, Cheng, Chen et al. (2009) reported that *Cyperus* and *Canna* roots grew at rates of 7,000 g/m^2 and 5,500 g/m^2, respectively.

Table 5. 7. Total dry root mass per plant treatment after the experiment (mean ± S.E., n = 9).

Treatments		IG [a]	CI [a]	CP [a]	CI-IG [a]	IG-CP [a]	CI-CP [ab]	CI-IG-CP [b]
Mean (g)		0.926	1.40	1.49	1.90	1.99	2.40	4.50
S.E. (± g)		0.246	0.47	0.35	0.64	0.50	0.81	0.73
Rank		Low	Low	Low	Low	Low	Medium	High
p-values	IG	--						
	CI	0.99	--					
	CP	0.99	1.00	--				
	CI-IG	0.88	0.99	0.99	--			
	IG-CP	0.83	0.98	0.99	1.00	--		
	CI-CP	0.53	0.87	0.91	0.99	0.99	--	
	CI-IG-CP	0.0008	0.0054	0.0074	0.032	0.043	0.14	--

Letters indicate statistical significant mean differences of root development between plant treatment at α = 0.05. *Iris* (IG), *Canna* (CI), *Cyperus* (CP), *Canna-Iris* (CI-IG), *Iris-Cyperus* (IG-CP), *Canna-Cyperus* (CI-CP), *Canna-Iris-Cyperus* (CI-IG-CP), and *control* (C).

For all treatments, infiltration rates were significantly higher during August 2015 than during the rest of sampling dates (Figure 5.14, p<0.0001). In general, there was a statistically significant interaction between time and treatment (p = 0.006). A substantial reduction in the infiltration rate from August to September 2015 was expected, based on previous studies showing rapid accumulation of solids in the surface of the filter (Hua, Zhu et al. 2010) and own observations of crust formation in this study. Then, infiltration rates continued to decrease slightly in subsequent months for all plant treatments.



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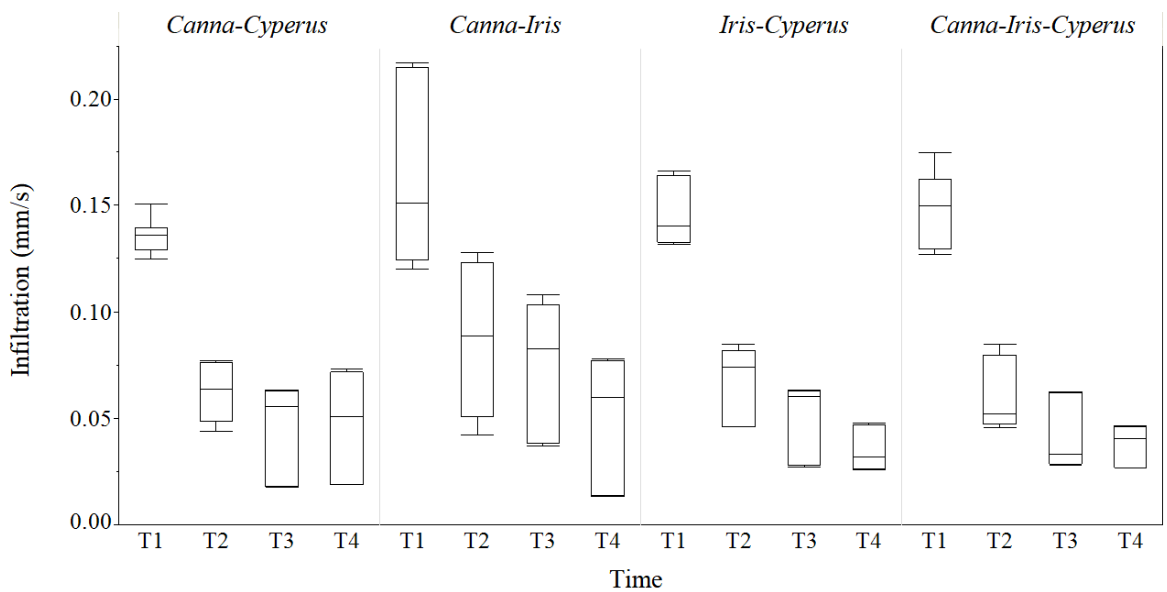

b

Figure 5. 14. a. Infiltration rates (median, n = 9) for *Canna*, *control*, *Cyperus*, and *Iris* based on time. b. Infiltration rates (median, n = 9) for *Canna-Cyperus*, *Canna-Iris*, *Canna-Iris-Cyperus*, and *Iris-Cyperus* based on time. T1: August 2015, T2: September 2015, T3: November 2015, and T4: February 2016.

Due to the destructiveness for collecting media core samples and root biomass, relationships for infiltration rates based on time, root biomass, and volatile solids were collected once at the end on the experiment. Overall infiltration rates (i.e., including all sampling times, Table 5.8) were the highest for the unplanted columns and columns planted with *Canna-Iris* and *Canna* plants. Remaining infiltrations rates, while lower than those observed in the control columns, were not significantly different from each other. In individual plant treatments, *Canna* plants did not decrease the infiltration rate significantly from the control (p = 0.847); however, infiltration rates for the *Canna* columns were significantly faster than those for columns planted with *Iris* (p = 0.042). The slowest infiltration rates, amongst all treatments, were observed for columns planted with only *Cyperus* (p = 0.001 versus the control).

Table 5. 8. Relative infiltration rates rankings for plant treatments (n = 36).

Treatment	**Mean infiltration rate (mm/s)**	**Overall**	**August 2015**	**September 2015**	**November 2015**	**February 2016**
Canna-Iris	0.0941 ± 0.0089	High	High	High-med	High-med	Med
Control	0.0917 ± 0.0091	High	High	High-med	High-med	Med
Canna	0.0908 ± 0.0063	High	Med	High	High	Med
Iris	0.0767 ± 0.0064	Low	Med	High-med-low	Med-Low	Med
Iris-Cyperus	0.0750 ± 0.0076	Low	High-med	High-med	Med-low	Med
Canna-Cyperus	0.0729 ± 0.0016	Low	Med	Med-low	Low	Med
Canna-Iris-Cyperus	0.0718 ± 0.0079	Low	High	Med-low	Low	Med
Cyperus	0.0699 ± 0.0064	Low	Med	Low	Med-Low	Med

Caption: Based on significance of 0.05, relative infiltration rates rankings for plant treatments of high, medium (med), and low for each sample time. Some treatments are ranked as high-med, med-low, or high-med-low to indicate there was no statistical difference between that specific treatment and other treatments in the high, medium, or low rankings. There was no statistical difference in infiltration rates in February 2016, therefore, all treatments are ranked as medium.

In general, control columns without plants were categorized as having high (or high and medium) infiltration rates at all sampling times; therefore, there was no significant evidence that plants increased infiltration in the column studies. On the contrary, columns with plants generally exhibited lower infiltration rates than the control columns, especially columns planted with *Cyperus* and *Canna-Cyperus*. However, as the experiment progressed, infiltration rates from all treatments converged, so that there were no significant differences between infiltration rates in the unplanted columns and the planted columns by February 2016.

Within the planted columns, columns with *Canna* plants had the highest infiltration rates. For example, volatile solid accumulation and root development were similar for *Canna*, *Canna-Iris*, *Cyperus*, and *Iris-Cyperus*; however, treatments with *Canna* had higher infiltration rates. *Canna* has a bulb type root with a horizontal propagation method of rhizome division (Cabi 2016), and from each division, shoot and root growth can promote the opening of pores in the filter media (Wang, Xu et al. 2008) (Figure 15.a). In contrast, lower infiltration rates were measured for columns with *Cyperus* (e.g., *Cyperus* and *Canna-Iris-Cyperus*), despite similar volatile solid accumulation and root development. *Cyperus* has a tough root with numerous rootlets that forms a dense culm at the surface of the soil (Cabi 2016) along with denser and longer root development from 5 to 15 cm deep of the filter media (Cheng, Chen et al. 2009). Thus, *Cyperus* roots could decrease infiltration (Figure 15.b).

Similar to *Cyperus*, *Iris* root has a rhizome apical root, from which fine auxiliary buds and adventitious roots expand (Iris 2016). However, *Iris* roots are thinner than *Cyperus* roots and occupy less void space in the filter media (Figure 15.c). Infiltration rates for columns planted with Iris were typically between rates for columns with *Canna* and *Cyperus*, particularly when *Iris* was combined with other plant, except all three plants.

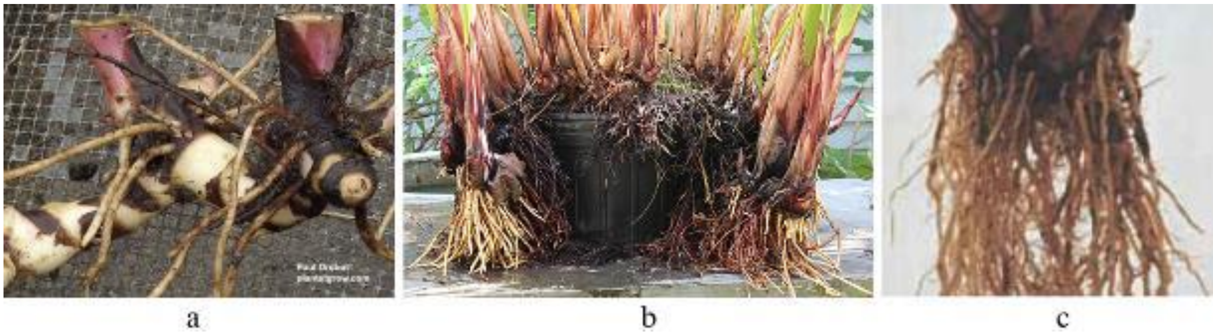

Figure 5. 15. a. Root structure for *Canna* (Picture was taken from www.plantsgrow.com); b. Root structure for *Cyperus* (Picture was taken from www.olabrisagardens.com); and c. Root structure for *Iris* (Picture was taken from www.grovida.us).

5.4. Conclusion

In operation since March 2013, the VSSF-CTW, as a part of a decentralized, self-sufficient, close-loop, organic waste treatment system in Costa Rica, has been effectively treating anaerobic digestate without incidences of clogging. Reduction of organic and solid loads by anaerobic digestion and mechanical separation, settling within the effluent storage tank, and filtration of fine solids by the geotextile membrane has likely prevented clogging within the VSSF-CTW. Additionally, only recirculated wastewater was applied to the main treatment area with the goals of increasing nutrient removal while maintaining a robust plant community. Under this operation, accumulation of fixed solids has not impacted the filter media, as demonstrated by the mass balance of the solids that estimated a constant void space of 20 m^3 in the main treatment area. Therefore, for future research and optimization, the use of a geotextile membrane is recommended for designs in Costa Rica. Moreover, there is a need to evaluate the dynamics of the solids during treatment in the filter media due to recirculation and flooding.

The filter media below the geotextile membrane was likely rapidly filled with solids, as demonstrated by the mass balance of the solids. However, this phenomenon did not appear to impact wastewater treatment; however, replacement of this portion of the treatment area could be necessary. Planting selection is not only important for nutrient removal but also for reducing incidence of clogging. The identification of morphological traits of root plants that reduce clogging is important for extending the lifespan of wetlands. In this study, low correlations were found when infiltration was analyzed as a function of volatile solids and root biomass in the sand media. More data is needed to determine impacts of volatile solids and root biomass within the full-scale wetland. Trends were identified; root growth of *Canna* plants did not adversely affect infiltration. In the full-scale wetland, infiltration rates near *Canna* plants were relatively high despite massive root development. Within the column study, columns planted with *Canna* had higher infiltration rates than columns planted with *Cyperus*, even though solid accumulation and root biomass were similar.

5.5. Acknowledgment

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CHAPTER 6: EXERGY-BASED ASSESSMENT OF SUSTAINABILITY OF A SOLAR-POWERED ANAEROBIC DIGESTION AND HYBRID CONSTRUCTED TREATMENT WETLAND SYSTEM TO TREAT AGRICULTURAL WASTES IN COSTA RICA

Abstract: Agriculture and ecological tourism are both crucial to Costa Rica's economy. However, agricultural activity can damage ecosystems when residues and wastewater are discharged onto land and into surface water. Integration of profitable bioenergy systems with engineered ecological treatment systems, such as the solar-powered anaerobic digestion and hybrid constructed treatment wetland (SPAD-HCTW), can potentially balance agricultural production and ecological protection. The SPAD-HCTW integrates solar heating, anaerobic digestion, and constructed treatment wetland technologies to treat biomass residues and agricultural wastewaters while producing energy, fertilizer, and treated water. In this study, an exergy-based assessment of sustainability using two indexes, the environmental exergy efficiency ($\eta_{env,ex}$) and the total pollution rate ($R_{pol,ex}$), was conducted. Calculations were completed for three different cases to determine how inclusion of constructed treatment wetlands affected exergetic sustainability. The baseline case was the solar-powered anaerobic digester (SPAD) alone. Two additional cases, considering either treatment of the digestate (1) by a vertical subsurface flow constructed treatment wetland (VSSF-CTW) or (2) by the VSSF-CTW in series with a free water surface constructed treatment wetland (FWS-CTW), were analyzed. Results indicated that the baseline case, which only provided energy and fertilizer, was not sustainable from an exergetic point of view ($\eta_{env,ex}$ = 0.469 ± 0.094 and $R_{pol,ex}$ = 1.13 ± 042). More exergetically favorable $\eta_{env,ex}$ and $R_{pol,ex}$ values were obtained with the inclusion of a VSSF-CTW; however, case 1 was still not considered exergetically sustainable ($\eta_{env,ex}$ = 0.734 ±

0.201 and $R_{pol,ex}$ = 0.362 ± 0.373). The entire SPAD-HCTW, or case 2, was exergetically sustainable with a $\eta_{env,ex}$ of 5.60 ± 1.56 and a $R_{pol,ex}$ of -0.821 ± 0.167, due to a positive exergy balance in which the food waste and chicken litter were converted into high quality end products (i.e., energy, fertilizer, and treated water). Through examination of these case studies, multiple approaches for calculation of the exergy of treated and stored wastewater were evaluated. Exergetically, this study shows that technical innovation in conversion of agricultural wastewater and biomass wastes into resources (e.g., fertilizer, biogas, and treated water) can help address the adverse impacts of agricultural production on ecosystems.

6.1. Introduction

Agriculture is an important economic driver in Costa Rica, contributing 9.19% to the total Gross Domestic Product (GDP) (EN 2015). However, while contributing to local economies, agricultural activities also consume 84% of Costa Rica's annual fresh water demand (12,320 m^3) and produce 6,000 tons per year of biomass wastes (GWP 2012, Coto 2013). Most agricultural wastewaters and biomass wastes are directly discharged into aquatic systems with limited or no treatment (Ruiz 2014), adversely affecting ecosystems that positively contribute to the Costa Rican economy and environment (EN 2015). In 2014, ecosystems contributed to $2.6 billion (5.3% of the GDP) to the Costa Rican economy through tourism alone (ICT 2014). Environmentally, ecosystems in Costa Rica sequestered 26 tons per hectare of carbon each year (Salazar 2014). However, while there is an economic need to protect ecosystems from the discharge of agricultural wastes, limited funding is allocated for water resource management and protection in Costa Rica. For example, it is estimated that Costa Rica should spend $2.0 billion to manage its water resources from 2010 to 2020 (MINAET 2010); however, actual financial commitments are substantially less than this (GWP 2012, Echeverria and Cantillo 2013). To confront these challenges, low-cost, sustainable systems that convert wastewater and biomass residues into valued resources would greatly benefit agricultural areas in Costa Rica.

Combining anaerobic digestion and constructed treatment wetland (CTW) technologies can efficiently treat biomass residues and wastewater, generate renewable energy, produce fertilizers, and reclaim water (Barros, Ruiz et al. 2008, Ruiz, Díaz et al. 2010, Comino, Riggio et al. 2013, de la Varga, Díaz et al. 2013). Comino, Riggio et al. (2013) reported that a mesophilic continuous stirred tank reactor (CSTR) followed by parallel VSSF-CTWs and an in-series horizontal subsurface flow constructed treatment wetland (HSSF-CTW) removed 88% of chemical oxygen demand (COD), 73% of nitrate (NO_3), 98% of ammonium (NH_4), and 99% of phosphate (PO_4) from a discharge of 0.200 m^3/d, with a maximum organic load rate of 160 g COD/m^2d. Barros, Ruiz et al. (2008) highlighted the benefits of pretreating municipal wastewater with an up-flow anaerobic sludge blanket (UASB) reactor receiving 3 m^3/d and an organic load of 4,000 mg COD/L. The UASB reactor reduced 80-90% of TSS, 65% of COD, and 40-50% of BOD, while the CTW removal efficiencies were 32-52% for total suspended solids (TSS), 83% for COD, and 87% for biological oxygen demand (BOD). Similarly, de la Varga, Díaz et al. (2013) and Ruiz, Díaz et al. (2010) found that an UASB reactor provided high TSS removal, while the CTWs reclaimed the water by reducing COD concentrations, which were still high after the anaerobic digestion pretreatment (COD > 150 mg/L).

Exergy is a thermodynamic concept that describes system performance according to the second law of thermodynamics that can be used for assessing the sustainability of systems. Opposite to entropy, which measures the low-quality energy of materials in thermodynamic disequilibrium, exergy measures the quantity and quality of energy that a particular material possesses if it is brought into thermodynamic equilibrium (Jørgensen 2006, Rosen 2012, Querol, Gonzalez-Regueral et al. 2013). This allows exergy to be a tool for resource accounting and a valuable sustainability metric (Chen, Chen et al. 2011). A system that converts materials with high entropy into high quality end products with low entropy will be sustainable if the balance between inputs (e.g.: wastes) and outputs (e.g.: biogas, fertilizers, and reclaimed water) is positive (Wall 2010, Woudstra 2016). Sustainability is a complex concept that varies broadly depending on the discipline and the context in which is evaluated. In this regard, a sustainable assessment based on exergy summarize only part of the environmental portion of sustainability as it evaluates how potential raw material is upgraded to valuable products, thereby alleviating negative environmental impacts of wastewater and biomass residues (Parsapour 2012).

Exergy analysis has been used previously to analyze the sustainability benefits of anaerobic digestion. Dong, Chi et al. (2014) found that implementing anaerobic digestion as a pretreatment step for sewage sludge composting significantly alleviated the environmental burden of sludge disposal. The environmental exergy efficiency ($\eta_{env,ex}$), or the ratio of the total exergy outputs of products over the inputs, was 16.9% for the composting process alone; inclusion of AD prior to composting increased $\eta_{env,ex}$ to 34.6% due to production of biogas (Dong, Chi et al. 2014). Likewise, Chatzipaschali and Stamatis (2015) indicated that $\eta_{env,ex}$ of a steam production plant increased when the biogas produced by anaerobic digestion of cheese whey was used for the thermal and electrical needs of the plant (Chatzipaschali and Stamatis 2015). Siefert and Litster (2014) modeled the exergetic effects of including an anaerobic digester with a solid oxide fuel cell (SOFC) into a conventional wastewater treatment plant. The resulting $\eta_{env,ex}$ was 58%, mainly due to the production of biogas to fuel the SOFC (Siefert and Litster 2014).

Other authors, e.g., Tang, Fang et al. (2014), Chen, Chen et al. (2011), and Shao and Chen (2015), have evaluated wetland ecosystems using exergy. Tang, Fang et al. (2014) used exergy, biomass, and diversity as indicators to evaluate the development and health status of macrofauna for a wetland planted with *Sonneratia apetala* (mangrove apple). Exergy increased from 2 MJ/m^2 to 9 MJ/m^2 in 1,200 days due to the fast growth of *S. apetela* (Tang, Fang et al. 2014). Chen, Chen et al. (2011) applied cosmic exergy to analyze three wastewater treatment systems. Cosmic exergy considers the cosmos as a background, as the thermodynamic equilibrium environment for all the processes in the Earth (Chen, Chen et al. 2011). Chen, Chen et al. (2011) determined the natural, construction and operational inputs, and the ecological service outputs of a CTW, an activated sludge system, and a cyclic activated sludge system. Authors found that the renewable dependency, or the total renewable input over the ecological input, for the CTW (67%) was more than double the renewable dependency of the activated sludge system (38%) and the cyclic activated sludge system (31%), indicating that the CTW was more sustainable and environmentally friendly (Chen, Chen et al. 2011). Finally, Shao and Chen (2015) used embodied cosmic exergy, which measures the resource utilization efficiency taking into consideration the renewability index of all inputs of a system. The renewability index is the total natural and purchased renewable resources over the total resource use in the studied system. Therefore, the embodied cosmic exergy analysis can be used to determine which inputs should be changed to improve a system's renewability. Based on this tool, Shao and Chen (2015) determined that use of inputs with higher renewability indexes (e.g. PVC pipes, 1.23% versus cement pipes, RI = 0.89%) during the construction and operation of a constructed wetland would increase the renewability index from 2.09% to a higher magnitude that was not indicated by the authors (Shao and Chen 2015).

Exergy analysis has also been used to quantify water quality. Tai and Matsushige (1986) determined that the exergy of COD in wastewater is 13.6 kJ per g COD (Tai and Matsushige 1986). Hellstrom (1997) applied Tai's ratio to compare energy and exergy while estimating the physical consumption of resources during conventional wastewater treatment. Martínez, Uche et al. (2010) evaluated the exergy efficiency of desalinization during wastewater treatment. Inputs to the treatment plant were wastewater, described by its COD and salt concentrations, silica used for coagulation and flocculation, and electricity. Outputs were COD, fat, salts, and silica in the discharged effluent and sludge. Authors found that chemical-based desalinization processes (multi-stage flash distillation, multiple effect distillation, reverse osmosis, and electrodialysis) were less energy efficient than pumping techniques as the ratio of inputs over outputs was much greater than 1 for chemical-based desalinization (Martínez, Uche et al. 2010). Khosravi and Panjeshahi (2013) used an exergy analysis as an optimization tool to identify losses of exergy at the different stages of the wastewater treatment by using two sustainability indexes, $\eta_{env,ex,}$ and total pollution rate ($R_{pol,ex}$, ratio of lost exergy from wastes to total exergy outputs). Khosravi and Panjeshahi (2013) were able to increase efficiency of wastewater treatment by identifying inefficient steps, increasing $\eta_{env,ex}$ from 0.14 (low efficiency) to 0.36 and decreasing $R_{pol,ex}$ from 6.14 (large pollution rate) to 1.79.

Based on previous technical performance analysis (Aguilar Alvarez, Bustamante Roman et al. 2016), it was demonstrated how each separate unit of the SPAD-HCTW performed as a close-loop system to yield energy, fertilizers, and treated water. Exergy metrics have successfully been used to document the sustainability of systems similar to components of the SPAD-HCTW; consequently, exergy metrics were chosen to analyze the sustainability of the SPAD-HCTW since the technical point of view. Therefore, an exergy-based assessment of sustainability using two indexes ($\eta_{env,ex}$ and $R_{pol,ex}$) was conducted. The exergy baseline of the system was the SPAD (case 0) alone. Further analysis considered the SPAD and the VSSF-CTW (case 1) and, finally, the SPAD, the VSSF-CTW, and the FWS-CTW in-series (case 2) to compare the potential improvement of sustainability through inclusion of wetlands during treatment of agricultural wastes.

6.2. Material and Methods

The SPAD-HCTW was designed and built at the Fabio Baudrit Experimental Station (EEAFBM), in Alajuela, Costa Rica (10.00 m N, -84.26 m W) during 2011 and 2012. Materials for construction of the SPAD-HCTW were bought at local suppliers in Costa Rica. The system began operation in March 2013. This study was conducted from August 2015 to March 2016.

6.2.1. System description

The SPAD-HCTW is shown at Figure 6.1 and is described as follow. Flat-plate solar thermal collectors heated water to 50-78^{o}C. Heated water was pumped with a solar heating fluid transfer pump (model UP 26-99 F from Grundfos, Olathe, KS) and stored in a 3-m^3 hot water tank. Then, a digester heating pump (model PB 351MA from Wilo, Korea) circulated the hot water through a heat exchanger within the anaerobic digester to maintain thermophilic conditions (i.e., 45 ± 2^{o}C). The anaerobic digester was fed with a mixture of food wastes and chicken litter at a rate of one cubic meter per day, from Monday to Friday. Each week, 5 m^3 of feed mixture was prepared using treated water from the FWS-CTW. First, 151 ± 17.1 kg of dry chicken litter was mixed with one cubic meter of treated water in a 50-gallon drum, and then pumped by feed preparation pump (model 50ut2.4s-61 from Tsurumi, Glendale Heights, IL) to the feeding tank.

Next, approximately 17.4 ± 3.26 kg of dry food waste was ground (model Leeson C 184K17FB150 from ICAFE ®, Costa Rica), mixed with 4 m^3 of treated water, and then pumped using the feed preparation pump to the feeding tank. In the feeding tank, mixture was mixed for 30 minutes per week by an external feeding tank pump (model AMT P/N 1626-305-00 from AMT, Royersford, PA). From Monday to Friday, one cubic meter (~ 1000 kg) of feed was conveyed by the external pump to the anaerobic digester. The same volume of liquid digestate left the anaerobic digester and passed to the solid/liquid separator by gravity. In the anaerobic digester, a submersible digester mixing pump (model 5763 from AMT, Royersford, PA) mixed the anaerobic digester's contents for 10 minutes each hour. Biogas produced from the digester was stored in a biogas storage bag (HDPE 60 m^3 from Viogaz ®, Costa Rica) prior to combustion by a generator (model B4T-5000 Bioflex from Branco®, Brazil). Electricity was used to power pumps and other pieces of equipment in the system to satisfy operational requirements. Solids and liquid digestate were separated mechanically with a rotary screen separator (ICAFE ®, Costa Rica, particle size > 0.5 mm in diameter), which operated 10 minutes per day. The solid digestate was used as a fertilizer for on-site farming uses. The liquid digestate was stored in the effluent tank. Finally, the liquid digestate, one cubic meter per day from Monday to Friday was delivered by gravity to the HCTW to be further treated.

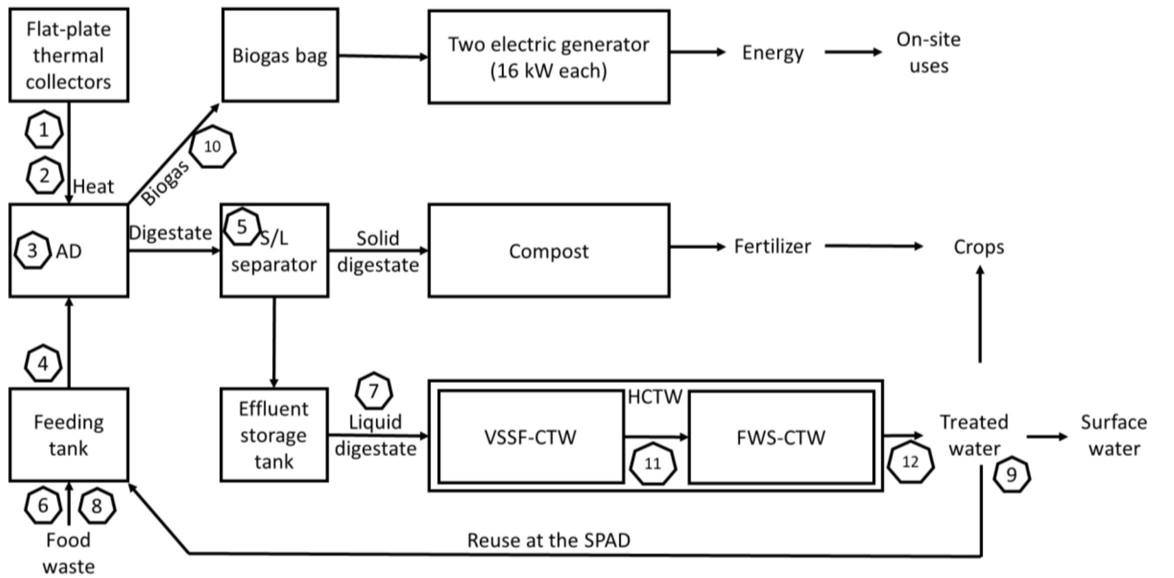

Figure 6. 1. Flowchart of the SPAD-HCTW system. Numbers indicate pumps and other equipment needed to operate the SPAD-HCTW: 1. Solar heating fluid transfer pump, 2. Digester heating pump, 3. Digester mixing pump, 4. Feeding tank pump, 5. Solid/liquid separator, 6. Grinder, 7. Effluent pump, 8. Feed preparation pump, 9. SPAD-HCTW exit pump, 10. Biogas flowmeter, 11. VSSF-CTW recirculation pump, and 12. FWS-CTW recirculation pump.

The HCTW consists of two CTWs in-series: a VSSF-CTW followed by a FWS-CTW (Figure 6.1). By gravity, the liquid digestate was discharged into the VSSF-CTW from the effluent tank. Then, from Monday to Friday, one cubic meter per day of the treated effluent from the VSSF-CTW was pumped to the FWS-CTW. The treated water from the FWS-CTW was used as either irrigation water or processing water for on-site uses. Both cells have the same dimensions. The top is 12.0 × 12.0 m, while the bottom is 9.00 × 9.00 m (median treatment area = 138 m^2). The height is 1.10 m and the slope of the walls is 27° with respect to the horizontal. The media profile of the VSSF-CTW is, from the bottom to top, 20 cm of stone (12 – 20 mm), 20 cm of pea gravel (4 – 8 mm), and 70 cm of coarse sand (0.75 – 2.0 mm, 32% porosity). The VSSF-CTW has a maximum water storage capacity in the media of 30.9 m^3, with an additional 80 m^3 of storage above the media. A 2.00 × 2.00 m geotextile membrane (GT 131 from Skaps, Athens, GA) was installed beneath the discharge of the pipe from the effluent tank to collect suspended sediments and prior to infiltration of the liquid digestate effluent into the media. The VSSF-CTW was planted with randomized blocks of one square meter of *Cyperus papyrus*, *Iris graminea*, or *Canna indica*. Total area covered by each plant was 10 m^2. The FWS-CTW has maximum water storage capacity of 152 m^3. The FWS-CTW was planted with two different species of floating plants: *Eichhornia crassipes* and *Pistia stratiotes*. Floating plants were divided by species by a frame of bamboo (3 m wide and 9 m length per species). The wetland unit includes three pumps: one recirculation pump installed at the bottom of each CTW (model WS V52 from Franklin Electric, Fort Wayne, IN) for recirculation within the same CTW and an exit pump in the FWS-CTW (model WS 102 from Franklin Electric, Fort Wayne, IN) that transferred the treated water to either the feeding tank or the irrigation system.

6.2.2. Boundary definitions

An exergy-based assessment of sustainability using two indexes was performed for three cases (Figure 6.2). Case 0, the baseline of the system, only considered the SPAD, which includes pieces of equipment from number 1 to 9 (Figure 6.1). The inputs included feed, electricity, and heat. The outputs were solid and liquid digestate, electricity from methane combustion by the generator, and carbon dioxide, both from the produced biogas. The produced biogas was used as the output for all cases, offset by different electrical usage due to additional pumps in Cases 1 and 2. Case 1 considered the SPAD and the VSSF-CTW. Water (rain and runoff) caught by the VSSF-CTW, and electricity needed for running the VSSF-CTW recirculation pump were added to the inputs for case 0. Outputs for case 1 were biogas, solid digestate, sediments, plant biomass, and treated effluent from the VSSF-CTW. Case 2 considered the entire SPAD-HCTW. The inputs for this case were the ones for case 1 and rain and runoff caught by the FWS-CTW, and electricity demand by the FWS-CTW recirculation pump. The corresponding outputs were biogas, solid digestate, sediments, plant biomass, and treated water from the FWS-CTW.

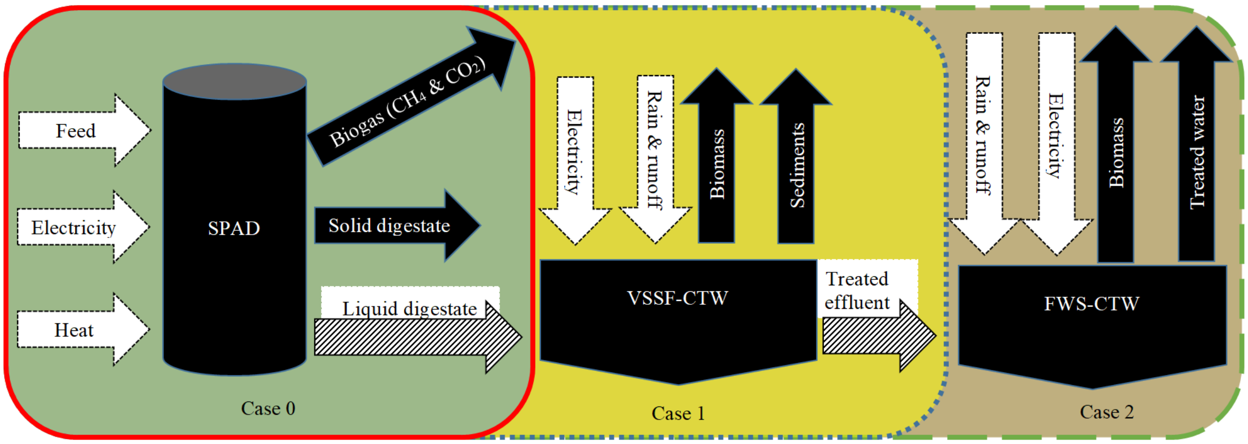

Figure 6. 2. Boundary definitions for conducting the exergy-based calculation of sustainability for the SPAD-HCTW. Case 0 – the SPAD. Case 1 – the SPAD with the VSSF-CTW. Case 2 – SPAD with the VSSF-CTW and FWS-CTW.

6.2.3. Data collection

The SPAD-HCTW had been running for 30 months as of August 2015. Data collected from August 2015 to March 2016 were used for this study – a time period which includes both phases of low and high precipitation. JMP®, Version 10.0.0 (SAS Institute Inc., Cary, NC) was used to classify precipitation data from August 2015 to March 2016. Weeks with precipitation from 0.00 to 1.91 cm/week were considered as absent to low precipitation period. This dry condition persisted from January to March 2016. On the other hand, weeks with precipitations higher than 1.91 cm/week were considered as medium to high precipitation period, which corresponded to a rainy period from August to December 2015. Data collection is detailed below for each case.

6.2.3.1. Case 0 – the SPAD

Feed consisted of chicken litter, food waste, and treated water. The chicken litter came from approximately 4,000 laying chickens at the experimental station. Food waste was transported from a nearby food distribution facility. Wastes mainly consisted of non-commercial over-ripe or damaged vegetables and fruits, including cucumbers, peppers, avocado, papayas, pineapples, and tomatoes. Water for preparing the feed came from the treated water produced by the FWS-CTW. Quantities of chicken litter, food wastes, and treated water were measured and recorded every week. Chicken litter and food wastes were weighed by an industrial scale (Romanas Oconi S.A., Costa Rica). The volume of treated water was estimated from pump time. Samples of the mixed feed were collected and analyzed for COD, TS, TN, and TP at the Water Quality Laboratory at the Research Center of Environmental Pollution at the University of Costa Rica (CICA-LCA); whereas, the chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B) of the feed was analyzed at the Agronomy Research Center at the University of Costa Rica laboratory (CIA).

Electricity usage for equipment was calculated based on duration of equipment operation. The equipment included in case 0 was seven pumps, all pumps but the VSSF-CTW and FWS-CTW recirculation pumps, a solid/liquid separator, and a grinder (Figure 6.1). Heat exergy input was calculated by recording, at a data acquisition unit (DAQ, model CR1000 Campbell Scientific, Logan, UT), the temperature from the hot water tank and the digester. Environmental temperature (°C) was recorded every 2 minutes by a weather station located on-site (IMN, 10.00 m N, -84.26 m W) and averaged for each hour. The total daily flow rate of hot water was calculated from the fixed flow rate of the hot-water pumps and pumping time. Heat from electrical generator was not considered as it was negligible when compared to the heat from the flat-plate solar collectors and would be relatively constant for all cases.

Biogas production was measured by a flowmeter (EKM-PGM 75 from EKM Metering, Santa Cruz, CA) and the total amount of biogas per week was recorded. Biogas samples were collected by using a sampling pump (SKC® Grab Air, Bag Sampler Cat. No. 222-2301) and analyzed monthly for methane and carbon dioxide by the Center for Research in Electrochemistry and Chemical Energy at the University of Costa Rica (CELEQ).

Solid digestate was separated by a solid/liquid separator, collected, and weighed by an industrial scale (Romanas Oconi S.A., Costa Rica) weekly. Particle size higher than 5 mm in diameter were separated by the screen. Total solids (TS) content of the solid digestate was also measured weekly. The chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B) of the solid digestate was analyzed at the CIA laboratory.

The amount of liquid digestate generated from the digester unit was estimated from pump time. Weekly samples were analyzed for COD, TS, TN, and TP. Since the liquid digestate still contains suspended sediments, the liquid digestate samples were dried, and the chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B) of the dried sample was analyzed at the CIA laboratory.

6.2.3.2. Case 1 – the SPAD with the VSSF-CTW

Case 1 included all inputs and outputs of case 0. The electricity used by the VSSF-CTW recirculation pump, calculated from duration of pump use, and water input by the rain and runoff were included as additional inputs (Figure 6.2). Electricity usage for the VSSF-CTW recirculation pump was calculated based on duration of equipment operation. Precipitation data was collected from a weather station located on-site (IMN, 10.00 m N, -84.26 m W). Runoff was calculated for the drainage area of 1,500 m^2, applying the curve number method (Ward and Trimble 2004). Harvested plants, suspended sediments collected by the geotextile, and treated effluent from the VSSF-CTW were the additional outputs for case 1. Three plants, *Cyperus papyrus*, *Iris graminea*, and *Canna indica*, were planted in the VSSF-CTW during installation in 2012. Reported growth rates for each plant are 4.06 ± 0.33, 6.76 ± 0.29, and 8.30 ± 0.69 $g/m^2/d$ on a dry mass basis (DM) for *Cyperus papyrus*, *Iris graminea*, and *Canna indica* , respectively (Li, Yang et al. 2013). Each month, one square meter of each plant was harvested, and weighed by an industrial scale (Romanas Oconi S.A.). Moisture content on the tissues of each plant was measured for obtaining the dry weight of the harvested plants. Plant samples were also analyzed at CIA for chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B). Suspended sediments caught by the geotextile were collected and weighed by an industrial scale (Romanas Oconi S.A.). Moisture content was measured to obtain the dry weight, and a sample was analyzed at CIA for determining its chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B). The effluent from the VSSF-CTW was analyzed weekly for COD, TS, TN, and TP concentrations to determine the exergy of the water. The volume of wastewater entering and exiting the VSSF-CTW was estimated from pump time. Additionally, the change of volume of the saturated portion of the VSSF-CTW was measured weekly by measuring the water level at the well pump.

6.2.3.3. Case 2 – SPAD-HCTW

Case 2 considered the entire SPAD-HCTW system. In addition to all inputs and outputs used for the case 1, the electricity used by the FWS-CTW recirculation pump and rain and runoff were included in the case 2 (Figure 6.2). Two aquatic plants (*Eichhornia crassipes* and *Pistia stratiotes*) grew in the FWS-CTW. Reported growth rates for *Eichhornia crassipes* and *Pistia stratiotes* are 5.25 ± 0.28 and 1.12 ± 0.05 kg DM/m^2week. A square meter of both plants was harvested monthly. The harvested plants were air dried and weighed prior to chemical composition (C, N, P, Ca, Mg, K, S, Fe, C, Zn, Mn, and B) analysis. The treated water from the FWS-CTW was analyzed following the same procedure for the liquid effluent from the VSSF-CTW in case 1. Similar methods to those in case 1 were used to quantify rain and runoff, harvested plants, flow, and volume of water within the FWS-CTW.

6.2.4. Analytical methods

Total solids (TS) for chicken litter, food waste, solid digestate, liquid digestate, and effluent from the VSSF-CTW and the FWS-CTW were determined by the Hach method #8276. Samples for TS were placed in disposable aluminum dishes (VWR®, catalog number 25433-008), dried for 24 hours in an oven (OV-12A from Blue M, East Troy, WI), and dishes were weighed in an analytical balance (Ohaus Corporation, Mexico). Moisture content for sediments collected by the geotextile and harvested plants followed previous Hach method #8276. COD (Hach method #8000), TN (method MAQA-40), and TP (method MAQA-1) were determined for the substrate, liquid digestate, and treated effluent from the VSSF-CTW and FWS-CTW. A DRB 200 reactor (Hach product #LTV082.53.40001) and a DR 900 multiparameter handheld colorimeter (Hach product #9385100) were used to digest and measure COD digestion vials (high range digestion vials from 0 to 1,500 mg COD/L, Hach kit). TN and TP analyses were conducted at CICA-LCA. CICA-LCA analyses are based on the Standard Methods for the Examination of Water and Wastewater (Rice and Bridgewater 2012). The method 5310 B (modified) was followed for TN by a combustion method (TOC-V CSH/CSN from Shimadzu, Columbia, MD), while the method 4500-P D was followed for TP by a UV-visual spectrophotometer (Evolution 600 from Thermo Scientific, Madison, WI). Chemical composition (C, N, P, Ca, Mg, K, S, Fe, Cu, Zn, Mn, and B) of the feed, solid digestate, solids dried from the liquid digestate, and plant biomass from CTWs were analyzed at the CIA laboratory. Method SC09-LSF-P06 followed the Dunas method to determine N and C using an autoanalyzer (Vario Cube from Elementar, Philadelphia, PA), while the Method SC09-LSF-P10 determine P, Ca, Mg, K, S, Fe, Cu, Zn, Mn, and B using a IPC-plasma atomic emission spectroscopy (Optima 8300 ICP-OES Spectrometer from Perkin Elmer, Spain). Chemical composition (CH_4 and CO_2) of biogas samples were determined at the CELEQ laboratory by using a gas chromatograph (model HP6890 Plus from Hewlett Packard, Littleton, CO) equipped with a thermal conductivity detector. The column temperature was set up at 250°C and argon was used as carrier gas. CICA-LCA, CIA, and CELEQ are laboratories accredited by the Costa Rican Accreditation Institute (ECA).

6.2.5. Exergy calculation

To conduct the exergy-based assessment of sustainability, it was necessary to calculate the chemical exergy of the feed, solid digestate, solids in the liquid digestate, and harvested plants. The general method used to calculate the chemical exergy was based on the work done by Parsapour (2012). Standard chemical exergy values of the different substances and compounds are based on numbers from Szargut, Morris et al. (1988) and Ayres, Masini et al. (2001). Equation [6.1] defines the exergy for individual substances,

$$E_{chn} = \Delta G_f + \sum_e n_e E_{chne} \quad [6.1]$$

where E_{chn} is the exergy of the substance *n* [J/kg], ΔG_f is Gibbs free energy of formation of the chemical compound [J/kg], n_e is the mole fraction of the e_{th} substance, and E_{chne} is the standard chemical exergy of the element *e* [J/kg] (Szargut, Morris et al. 1988, Szargut 1989). However, in our case was needed to calculate the exergy for or a mixture of substances such as feed, the solid digestate, solids in the liquid digestate, and harvested plants. Equation 6.2 defines the exergy for a mixture of substances,

$$E_{chn,mixture} = \sum_i n_e E_{chne} + RT_0 y_e ln y_e \qquad [6.2]$$

where E_{chne} is the exergy of the substance *e* pure element of the substance [J/kg], *R* is the gas law constant [J/kg K], T_0 is temperature [k], and y_e is the mole fraction of the e_{th} substance (Parsapour 2012).

The physical exergy of the heat was calculated using equation 6.3,

$$E_{ph} = (h - h_0) - T_0(s - s_0) \qquad [6.3]$$

where E_{ph} is the physical exergy, *h* is the specific enthalpy [J/kg], h_0 is the specific enthalpy of saturated liquid water at room temperature [J/kg], *s* is the specific entropy [J/kg K], s_0 is the specific entropy of saturated liquid water at room temperature [J/kg K], and T_0 is the room temperature [K] (Martin and Parsapour 2012).

The exergy of water in the feed, liquid digestate, rain, runoff, treated effluent, and the reclaimed water were calculated from the standard chemical exergy of the water is 0.05 kJ/g (Martin and Parsapour 2012). For comparison, the exergy of the feed and liquid effluent was measured considering the COD, TN, and TP concentrations. The standard chemical exergies were 13.6 kJ/g for COD, 17.8 kJ/g NH_4-N for TN, and 1.40 kJ/g PO_4^{2-} for TP (Hellstrom 1997, Martínez, Uche et al. 2010).

The exergy coefficient of electricity is 1, meaning that "*1 kJ of electrical energy corresponds to an exergy flow of 1 kJ*" (Ayres, Masini et al. 2001), while the exergy of the biogas was calculated from the mass of methane and carbon dioxide. Methane was used for energy generation at a rate of 55.5 kJ per g of methane, and an overall methane utilization efficiency of 90% (Aguilar Alvarez, Bustamante Roman et al. 2016). The density of carbon dioxide is 1.84 kg/m^3, with a corresponding chemical exergy of 0.451 kJ/g.

The exergy balance was completed for cases 0, 1 and 2. Net exergy outputs and inputs were used to compare the performance of these three cases based on the environmental exergy efficiency ($\eta_{env,ex}$) and total pollution rate ($R_{pol,ex}$) sustainability indexes, which are defined as

$$\eta_{env,ex} = \frac{outputs_{ex}}{inputs_{ex}} > 1 \qquad [6.4]$$

$$R_{pol,ex} = \frac{inputs_{ex} - outputs_{ex}}{outputs_{ex}} < 0 \qquad [6.5]$$

Large values of $\eta_{env,ex}$ indicate little wasted energy and large quantities of high quality end products relative to inputs; therefore, high $\eta_{env,ex}$ values indicate little environmental degradation (Khosravi and Panjeshahi 2013). Large values of $R_{pol,ex}$ indicate low removals of pollutants when converting raw material into high quality end products (Khosravi and Panjeshahi 2013).

Labor is calculated in extended exergy accounting analysis as a factor to measure the exergy required to produce a good or service (Sciubba 2011). For the SPAD-HCTW, calculations of sustainability with and without exergy due to labor were calculated to examine its impact on the exergy balance. The exergy equivalent of labor is 78.7 MJ/h for Mexico (Sciubba 2011). This value was selected for the SPAD-HCTW, as Costa Rica and Mexico have similar GDP/person (Mexico is ranked 67, Costa Rica 77) and similar human development index (HDI), with Mexico in the 74th position and Costa Rica in the 69th.

6.2.6. Statistical analysis

Statistical analysis of data was conducted using JMP®, Version 10.0.0 (SAS Institute Inc., Cary, NC). Inputs and outputs consisted of 25 sampling events, 16 from August to December 2015 and 9 of them from January to March 2016. Data from inputs or output during the rainy or dry season were statistically compared at an alpha value of 0.05. Normality of the data was evaluated using the Shapiro-Wilk normality test ($p < 0.05$). The t-tests were conducted in case of normality. Otherwise, the Mann-Whitney test was conducted on data that failed the Shapiro-Wilk test.

6.3. Results and discussion

6.3.1. Case 0 – the SPAD

The total input exergy for the case 0 was 3,730 ± 402 MJ/week (Table 6.4). The feed contributed to 70.4% of the total inputs, as compared to heat (21.1%) and electricity (8.51%). Two approaches to estimate the exergy of the feed were compared. First, the exergy of the feed was calculated using equation 6.2, the elemental composition of the feed, and the standard chemical exergy of the water used for preparing the feed. The chemical composition of the feed consisted of C, N, P, and other minor components (Table 6.1). Per week, 169 ± 18 kg of dry matter of chicken litter and food waste were mixed with 4,611 ± 183 kg/week of water and the estimated input exergy was 2,351 ± 218 MJ/week. The second approach for calculating the exergy of the feed used previously reported chemical exergies of COD, TN, and TP. The estimated exergy was 2,624 ± 230 MJ/week, due to the large concentrations of COD, TN, and TP in the feed (Table 6.1). There was a difference of approximately 10% between the two approaches. This difference was 273 ± 448 MJ/week, indicating the standard error of the estimate exceeded the difference between the two methods. The second approach was used in this study as it had a larger value than the first one and was simpler to calculate. For the first approach, two important elements, Oxygen (O) and Hydrogen (H), were not included. Compared to carbon (34.2 kJ/g), oxygen has very low amount of exergy per gram (0.120 kJ/g) (Parsapour 2012), so no substantial changes would be expected if were included. Hydrogen, similar to Carbon, is important for biogas production and it has high amount of exergy (236.1 kJ/g) (Szargut, Morris et al. 1988, Ayres, Masini et al. 2001). Thus, hydrogen could substantially change the exergy values depending on the amount of hydrogen is in the analyzed substance. Parsapour (2012) estimated 7% of hydrogen in the raw effluent from a brewing process; whereas, after anaerobic digestion, the liquid digestate contained 4% of hydrogen. Considering these hydrogen percentages, the total exergy input in case 0 increased more than two times; however, the sustainable indexes ($\eta_{env,ex}$ and $R_{pol,ex}$) followed the same trend for each case. The SPAD alone would not be sustainable, as well as the SPAD plus the VSSF-CTW. Finally, the SPAD-HCTW would be sustainable.

A direct electric input of 88.2 ± 2.5 kWh/week was needed to run the equipment for case 0 (Table 6.2). The digester mixing pump had higher electricity consumptions than other equipment. The high electricity demand was caused by the larger size of the pump and longer running times that were required to fulfill the mixing demand (Table 6.2). The remaining pieces of equipment only consumed 29.0% of total energy demand. As 1 kJ of electrical energy corresponds to an exergy flow of 1 kJ, the total physical exergy input by electricity of the case 0 was 317 ± 8 MJ/week. The exergy due to electricity could be reduced by optimizing the operation of the different pieces of equipment at the case 0. For example, if the digester mixing pump only operates 5 minutes per hour (instead of 10 minutes per hour), the electricity consumption would decrease to 56.8 ± 2.5 kWh/week, and accordingly, the input exergy due to electricity (204 ± 2 MJ/week), which is 36% less than the current exergy. However, reducing mixing time would impact methane production and treatment performance of wastes. Wang and Larson (2015) observed that digesters with no and intermittent mixing had significantly lower methane production rate (< 1.5 L methane per kg VS destroyed) than a continuously mixed anaerobic digester (> 2.0 L methane per kg VS destroyed).

The total physical exergy of heat provided by the solar panels was calculated as the sum of heat transferred from the solar panels to the hot water storage tank and then to the digester. The solar heating fluid transfer pump operated 7 hours per day, with a flow rate of 0.960 m^3/h. The amount of water used to transfer heat from the solar panels to the hot water storage tank was 47,040 kg/week. The average temperature of the hot water storage tank was 68.6 ± 3.2°C. The digester heating pump ran 43.9 ± 15.7 h/week at a flow rate of 0.720 m^3/h. In total, 31,624 kg/week of hot water at 54.8 ± 5.8°C were needed to heat the digester. The corresponding physical exergy of the heat was calculated by Equation 6.3, considering a local environmental temperature of 25°C. The total heat exergy input was 787 ± 162 MJ/week.

Table 6. 1. Inputs and outputs chemical characterization.

	Case 0														
	COD	TN	TP	C	N	P	Ca	Mg	K	S	Fe	Cu	Zn	Mn	B
	(mean ± SD, n = 28, mg/L)			(n = 3, %)											
Feed	38,000 ± 2,700	860 ± 40	279 ± 25	36	4.5	1.3	5.0	0.92	4.8	0.82	0.060	0.00	0.010	0.020	0.010
Solid digestate	--	--	--	41	2.5	2.1	5.4	0.51	0.41	0.59	0.15	0.010	0.020	0.030	0.00
Liquid digestate	6,841 ± 681	1,000 ± 100	106 ± 16	26	2.9	1.7	5.3	1.3	10	0.70	0.08	0.010	0.030	0.020	0.010
	Case 1														
				C	N	P	Ca	Mg	K	S	Fe	Cu	Zn	Mn	B
				(n = 3, %)											
Sediments collected by geotextile	--	--	--	26	2.9	1.7	5.3	1.3	10	0.70	0.08	0.010	0.030	0.020	0.010
C. papyrus	--	--	--	42	1.6	0.10	0.61	0.18	2.8	0.16	0.020	0.00	0.00	0.020	0.00
C. indica	--	--	--	34	4.7	0.45	0.45	0.54	6.8	0.25	0.030	0.00	0.010	0.46	0.00
I. graminea	--	--	--	46	0.95	0.15	0.92	0.13	1.8	0.11	0.030	0.00	0.00	0.010	0.00
	Case 2														
Eichhornia crassipes	--	--	--	39	1.9	0.12	1.8	0.41	3.9	0.15	0.010	0.00	0.00	0.030	0.00
Pistia stratiotes	--	--	--	37	2.0	0.16	4.4	0.60	3.3	0.23	0.080	0.00	0.00	0.040	0.010

Table 6. 2. Energy consumption per piece of equipment in the SPAD-HCTW.

Equipment	Power (hp)	Time (h/week)	Energy (kWh/week)	Schedule
Solar heating fluid transfer pump	0.17	49	6.0	7 hours/day, 7 days/week
Digester heating pump	0.46	43	15	6.27 hours/day, 7 days/week
Digester mixing pump	3.0	28	62	10 min/hour, 24/7
Feeding tank pump	2.0	0.5	0.75	30 min/week
Solid/liquid separator	1.0	0.85	0.63	10 min/day, 5 days/week
Grinder	5.0	0.50	1.8	30 min/week
Effluent pump	0.50	0.85	0.32	10 min/day, 5 days/week
Feed preparation pump	0.50	0.21	0.080	12 min/week
SPAD-HCTW exit pump	1.0	1.0	0.75	1 hour/week
VSSF-CTW recirculation pump	0.50	16.6	6.2	2.4 hours/day,7 days/week
FWS-CTW recirculation pump	0.50	16.5	6.1	2.3 hours/day,7 days/week

Exergy of the solar-provided heat varied with changes in the environmental temperature (Hellstrom 1997). During this study, a daily average temperature of 22.9 ± 0.1°C, with daily temperatures ranging from 19.3 to 25.4°C occurred. Hourly, temperatures range from 15.0 to 33.0°C. On the lowest daily temperature, 144.8 ± 26.5 MJ/d would be required to heat the digester, as compared to 112.5 ±23.2 MJ/week on the highest daily temperature. Much greater variations in the exergy required to heat a thermophilic digester would be expected in temperate climates, as large differences in temperatures occur between winter and summer. In this case, even though the t-test for two independent variables indicated that hourly temperatures from January to March 2016 (23.5 ± 0.1°C) were significantly higher ($p = 5.98 \times 10^{-14}$) than the ones from August to December 2015 (22.6 ± 0.1°C), the change in exergy input due to heat would not substantially impact the balance between inputs and outputs. The approach used herein to calculate the exergy used to maintain the digester temperature did not account for the heat loss from pipes and the hot water storage tank, as the loss would be offset by solar heat, resulting in a net zero impact on the exergy balance.

After digestion, the feed was upgraded to electricity (from biogas combustion) and solid digestate, with a total output exergy of 1,749 ± 161 MJ/week (Table 6.4). The electricity represented 99.6% of the total outputs as exergy, while solid digestate was only 0.369%. Average biogas production of the SPAD was 73.2 ± 6.7 m^3/week with methane contents of 66.5 ± 1.4% (v/v). The remainder of the biogas was mainly carbon dioxide (33.5 ± 1.4% (v/v)), which was considered a loss. Solids collected from solid/liquid separation were 0.467 ± 0.112 kg dry matter per week, which were upgraded to fertilizer, yielding an exergy gain of 6.46 ± 1.67 MJ/week for the baseline case, based on the chemical composition of the solid digestate (Table 6.1).

Liquid digestate and carbon dioxide were considered losses, with a total of 607 ± 76 MJ/week (Table 6.4). The SPAD discharged 4,578 ± 178 kg/week of liquid digestate that had very high concentrations of COD, TN, and TP (Table 6.1). Even though land application of liquid digestate is often used, this practice was avoided in this study as direct discharge to crops or water bodies could impair the environment (Martínez, Uche et al. 2010), especially given the close proximity of crop fields to surface waters at the research site. Thus, the liquid digestate was considered a loss. In total, the liquid digestate contained 21.5 ± 0.8 kg/week of dried solids. The exergy for the liquid digestate totaled to 459 ± 17 MJ/week when standard exergies based on elemental composition were considered. On the other hand, exergy of the liquid digestate was 587 ± 74 MJ/week when literature-values of chemical exergies for COD, TN, and TP were used to estimate the liquid digestate exergy (Table 6.4). Similar to the feed, this generalized approach based on COD, TN, and TP exceeded estimate based on chemical composition by 21.8%. The difference between the approaches was 128 ± 91 MJ/week, indicating that the difference between approaches was not substantial. Similar to the feed, the exergy based on COD, TN, and TP standard chemical exergies was slightly larger, and due to the lack of information, especially the hydrogen percentage in the liquid digestate, the exergy obtained by the COD, TN, and TP concentration was considered in the exergy balance. Finally, carbon dioxide was considered loss due to its global warming potential.

In case 0, $\eta_{env,ex}$ (0.469 ± 0.094) was low and the $R_{pol,ex}$ (1.13 ± 0.42) was large (Table 6.5). The low $\eta_{env,ex}$ was due to the substantially large input exergy of the feed, which was not completely converted to electricity; instead approximately 22.4% of the exergy in the feed was discharged as liquid digestate. Likewise, the large $R_{pol,ex}$ indicated that a relatively small percentage of the feed was converted into high quality end products. Even if the liquid digestate and its sediments were considered useful as fertilizer by land application, the $\eta_{env,ex}$ would only increase to 0.627 ± 0.130, which, while improved, does not indicate sustainability from an exergetic perspective. Labor had a minimal impact on exergy in Case 0 (Table 6.4 and 6.5). Labor included 2.5 hours per week to operate the equipment. If labor (196 MJ/week) were considered, the $\eta_{env,ex}$ and the $R_{pol,ex}$ (Table 6.5) did not change substantially, decreasing $\eta_{env,ex}$ by 5.01% and increasing $R_{pol,ex}$ by 9.93%.

More exergetically favorable $\eta_{env,ex}$ and $R_{pol,ex}$ values were obtained from January to March 2016 than from August to December 2015 (Table 6.5). The Mann-Whitney test for two independent samples indicated that there were not significant differences between inputs based on wet versus dry period. For example, composition of the feed (as COD, TN, and TP) between the wet and dry period was not significant different ($p = 0.329$). Table 6.3 shows that from January to March 2016, volatile solids (VS) were 64.5 ± 3.2% of the TS in the feed, whereas from August to December 2015 the VS were 61.5 ± 3.8% of the TS, indicating no significant difference based on season in the percent VS in the feed ($p = 0.887$). Additionally, the carbon to nitrogen ratio (COD/N) in the feed was higher, but not significantly ($p = 0.170$), from January to March 2016 (COD/N = 56.4 ± 8.4) than from August to December 2015 (C/N = 44.3 ± 3.0). However, the power of the statistical comparison for each parameter in Table 6.3 was low, indicating that it is possible that a difference may have been detected with a greater sample number. Despite the similarity in inputs, higher concentrations of methane were observed in the biogas during the dry period, as methane composition in the biogas was higher (70.0 ± 2.5% v/v) than during the wet period (64.4 ± 1.0% v/v; $p = 0.0314$). However, the total methane produced, and subsequent electricity production, did not statistically differ ($p=0.365$).

Table 6. 3. SPAD performance during the rainy and dry period.

Parameters	August to December 2015	January to March 2016	p value*	Power
Food waste	Cucumbers, peppers, avocados, pineapples, and tomatoes	Cantaloupes, watermelons, and papayas	--	--
C/N ratio (g COD /g TN)	44.3 ± 3.0	56.4 ± 8.4	0.170	0.352
VS/TS in the feed (%, g VS / g TS)	61.5 ± 3.8	64.5 ± 3.2%	0.952	0.084
Biogas composition (% CH_4)	64.4 ± 1.0 (n=5)	70.0 ± 2.5 (n=3)	0.0314	0.471
Electricity produced (MJ/week)	1,606 ± 226	1,925 ± 214	0.365	0.162

* Mann-Whitney test for two independent samples.

The sample numbers are 16 for August to December 2015 and 9 for January to March 2016, unless otherwise indicated.

6.3.2. Case 1 – the SPAD with the VSSF-CTW

Case 1 included the VSSF-CTW to treat liquid digestate from the SPAD. Total exergy input for the case 1 was 4,852 ± 815 MJ/week (Table 6.4). Electricity to run the VSSF-CTW recirculation pump (Table 6.2) and precipitation and runoff were the additional inputs. Electricity represented only 0.639%, while precipitation and runoff represented 22.7% of the inputs. Consequently, optimization of the operation of the VSSF-CTW recirculation pump did not represent a substantial reduction in exergy input. While input of water by precipitation cannot be avoided, better isolation of the wetland could have excluded more runoff. Runoff only occurred during the rainy period, and the 0.40 m tall berms excluded runoff from the wetlands 51% of the wet period days. If all runoff were excluded, the input exergy by rain and runoff would decrease from 1,100 ± 423 MJ/week to 260.7 ± 94.3 MJ/week, consequently, decreasing the total input for case 1 to 4,013 ± 486 MJ/week.

Total exergy output for case 1 was 3,561 ± 377 MJ/week (Table 6.4). Outputs from case 0 (i.e., methane and solid digestate) corresponded to 49.1% of the outputs from case 1. The liquid digestate, considered loss for case 0, was treated by the VSSF-CTW. The liquid effluent from the VSSF-CTW accounted for 44.3% of the exergy outputs for case 1.

During the rainy period, the VSSF-CTW removed 87.6% of TS, 99.0% of COD, 96.8% of TN, and 99.2% of TP from the liquid digestate, with final effluent concentrations of 543 ± 63 mg TS/L, 66.5 ± 12.8 mg COD/L, 34.0 ± 8.5 mg TN/L, and 0.801 ± 0.131 mg TP/L. With the exception of TS, the effluent met the surface water discharge limits for Costa Rica (TS < 50 mg/L, COD < 150 mg/L, TN < 50 mg/L, and TP < 8 mg/L) (MINAE-MSP 2007). The COD concentrations and the VS to TS ratios (29.3 ± 3.9% g/g) in the effluent were relatively low, indicating that solids in the effluent from the VSSF-CTW was mostly of mineral origin. Previous studies have indicated that mineral sediments in wastewater discharged into land or aquatic ecosystems, have minimal impact (Avila, Salas et al. 2013); consequently, the effluent would be expected to have minor impacts on ecosystems if discharged to aquatic ecosystems. However, other uses of the effluent were prioritized over surface discharged. First, effluent from the VSSF-CTW could be used for dilution of feed for the digester, as the reductions in COD, TN, and TP were greater than 95%. Additionally, the remaining effluent from the VSSF-CTW was also considered an output as it met standards for use as irrigation water with regards to COD, TN, and

TP. As land application of liquid manures is a common practice in Costa Rica, TS concentrations were not considered a barrier to use of the effluent for irrigation. Liquid manure, which is classified as manure with no more than 4% TS, has been measured to have concentrations of 23.9 g TS/L (Lorimor, Powers et al. 2004). The TS concentration applied is 44 times larger than the concentration from the VSSF-CTW effluent in this study; therefore, no adverse impacts from solids in the effluent on agricultural production are expected. Given the mineral nature of the solids, they would also not likely represent a source of nutrients in runoff from the fields after irrigation.

In contrast, during the dry period, only TP (1.88 ± 0.76 mg/L) met discharge limits for Costa Rica, as final effluent concentrations were 564 ± 120 mg COD/L, 2,926 ± 272 mg TS/L, and 246 ± 28 mg TN/L. However, due to dry conditions, this effluent was stored for dilution of the feed for continued operation of the SPAD. Consequently, the treated effluent from the VSSF-CTW was considered an upgraded product, as the effluent from the VSSF-CTW can be used for any purpose during the rainy months (e.g., irrigation or reuse at the SPAD) or during the dry period (e.g., reuse at the SPAD). For this reason, its exergy value was estimated by considering the standard chemical exergy of the water. As the effluent from the VSSF-CTW was upgraded, the estimation of the exergy based on COD, TN, and TP standard chemical exergies was low, 79.1 ± 15.4 MJ/week, which was only 5.01% of the exergy calculated with the standard chemical exergy of the water. Because the effluent was meant for irrigation or reuse at the SPAD, the exergy value due to COD, TN, and TP was considered an output, but only would increase net exergy due to the treatment of water at the VSSF-CTW to 1,657 ± 215 MJ/week for case 1.

The geotextile located at the inlet area of the VSSF-CTW enhanced treatment performance by providing solid-liquid separation. The geotextile, with apparent pore size of 0.30

mm, flow-through rate of 0.102 $m^3/s/m^2$, and permittivity of 2.2 s^{-1}, retained sediments in the liquid digestate. Sediments in the liquid digestate, considered losses for case 0, were collected and upgraded to fertilizer through composting for case 1 (and 2). On average, 6.46 ± 1.31 kg/week of dry sediments were retained by the geotextile, representing an exergy of 69.2 ± 1.9 MJ/week, based on chemical composition (Table 6.1).

Table 6. 4. Exergy-based assessment of sustainability for the SPAD-HCTW.

Case 0		**Case 1**		**Case 2**	
Input	**Exergy (MJ/week)**	**Input**	**Exergy (MJ/week)**	**Input**	**Exergy (MJ/week)**
Feed	2,624 ± 230	Case 0	3,730 ± 402	Case 1	4,852 ± 815
Electricity	317.4 ± 8.9	Electricity	22.24 ± 1.49	Electricity	22.13 ± 1.50
Heat	787.8 ± 162.3	Precipitation and runoff	1,100 ± 423	Precipitation and runoff	1,100 ± 423
Total	3,730 ± 402	Total	4,852 ± 815	Total	5,974 ± 1,239
Labor	196	Labor	353	Labor	432
Total input	3,926	Total input	5,205	Total input	6,406
Output	**Exergy (MJ/week)**	**Output**	**Exergy (MJ/week)**	**Output**	**Exergy (MJ/week)**
Solid digestate	6.461 ± 1.666	Case 0	1,749 ± 161	Case 1	3,561 ± 377
Electricity	1,743 ± 159	Water: *Irrigation* *Dilution* *Stored*	1,578 ± 200 -- -- *1,578 ± 200*	Water: *Irrigation* *Dilution* *Stored*	4,480 ± 601 *1,223 ± 528* *230.5 ± 9.0* *3,026 ± 64*
--	--	Sediments	69.20 ± 1.9	--	--
--	--	Plant biomass: *Harvested* *Unharvested*	164.1 ± 13.5 *4.135 ± 0.470* *160.5 ± 13.4*	Plant biomass: *Harvested* *Unharvested*	25,416 ± 1,411 *943.4 ± 49.2* *24,473 ± 1,362*
Total output	1,749 ± 161	Total output	3,561 ± 377	Total output	33,458 ± 2,389
Losses	**Exergy (MJ/week)**	**Losses**	**Exergy (MJ/week)**	**Losses**	**Exergy (MJ/week)**
Liquid digestate	587.4 ± 73.8	CO_2	20.39 ± 1.86	CO_2	20.39 ± 1.86
CO_2	20.39 ± 1.86	--	--	--	--
Total losses	607.7 ± 75.7	Total losses	20.39 ± 1.86	Total losses	20.39 ± 1.86

n = 28 for each input, output, and loss.

Table 6. 5. Exergy indexes for sustainability.

Parameters	Case 0		Case 1		Case 2	
	Index	Value	Index	Value	Index	Value
Base case studies	$\eta_{env,ex}$	0.469 ± 0.094	$\eta_{env,ex}$	0.734 ± 0.201	$\eta_{env,ex}$	5.60 ± 1.56
	$R_{pol,ex}$	1.13 ± 0.42	$R_{pol,ex}$	0.362 ± 0.373	$R_{pol,ex}$	-0.821 ± 0.167
Including labor	$\eta_{env,ex}$	0.446 ± 0.086	$\eta_{env,ex}$	0.684 ± 0.179	$\eta_{env,ex}$	5.22 ± 1.38
	$R_{pol,ex}$	1.24 ± 0.43	$R_{pol,ex}$	0.462 ± 0.383	$R_{pol,ex}$	-0.809 ± 0.166
Rainy period	$\eta_{env,ex}$	0.433 ± 0.111	$\eta_{env,ex}$	0.722 ± 0.214	$\eta_{env,ex}$	4.65 ± 1.36
	$R_{pol,ex}$	1.31 ± 0.59	$R_{pol,ex}$	0.385 ± 0.411	$R_{pol,ex}$	-0.785 ± 0.171
Dry period	$\eta_{env,ex}$	0.627 ± 0.144	$\eta_{env,ex}$	0.896 ± 0.225	$\eta_{env,ex}$	9.37 ± 1.81
	$R_{pol,ex}$	0.596 ± 0.367	$R_{pol,ex}$	0.115 ± 0.279	$R_{pol,ex}$	-0.893 ± 0.157
CapEx* ($)**	29,500		36,500		40,000	
Net energy (MJ/week)**	1,521		1,499		1,476	

*CapEx: Capital expenditure

**CapEx and Net energy values from Aguilar Alvarez, Bustamante Roman et al. (2016)

The VSSF-CTW provided the conditions (e.g., water, nutrients) suitable for plant growth; subsequently, the harvested biomass accounted for an exergy output. Chemical exergies for *Cyperus papyrus* (103,653 kJ/kg), *Iris graminea* (15,993 kJ/kg), and *Canna indica* (220,875 kJ/kg) differed due to the chemical composition of their tissues (Table 6.1). It was observed that *Canna indica* possessed higher chemical exergy than *Cyperus papyrus* and *Iris graminea* due to the amount of potassium (K) in the tissues. Potassium has a higher standard chemical exergy (9,400 J/g) than the other analyzed elements. Carbon also has higher standard chemical exergy (34,188 J/g), but the composition of the plants was similar. Net exergy output from harvested biomass totaled to 4.13 ± 0.47 MJ/week (Table 6.4). Non-harvested plants accumulated a biomass of 277 ± 23 g/week for *Cyperus papyrus*, 473 ± 20 g/week for *Iris graminea*, and 581 ± 48 g/week for *Canna indica*. This mass stored at the VSSF-CTW represented an output exergy of 160 ± 13 MJ/week. Similarly to Tang, Fang et al. (2014), plants at the VSSF-CTW were considered outputs as increased habitat for biodiversity and converted nutrients in the wastewater into an organized structure in the plants. Opio, Jones et al. (2014) and Zhang, Rengel et al.

(2007) have reported different growth rates as the ones reported by Li, Yang et al. (2013). In Uganda, a wetland planted with *Cyperus papyrus* had a productivity growth rate from 16.7 to 37.4 g/m^{2}d (Opio, Jones et al. 2014), whereas 33.5 ± 0.3 g/m^{2}d of *Canna indica* were measured at wetland microcosms in Australia (Zhang, Rengel et al. 2007). Results by Li, Yang et al. (2013) were used in this study to be conservative and avoid huge variabilities of growth rates obtained at field scale. The conservative assumption of plant growth rates in this study did not substantially impact the exergy output from biomass. When the less conservative rates were used for biomass growth for the exergy calculations, the relative output exergy of harvested biomass was 18.1 ± 6.2 MJ/week and plant biomass was 722 ± 1 MJ/week, increasing the output exergy for case 1 by only 16.0%.

In case 1, the $\eta_{env,ex}$ increased to 0.734 ± 0.201 and the $R_{pol,ex}$ decreased to 0.362 ± 0.373 (Table 6.5). Sustainability indexes changed in a positive way with respect to case 0 due to inclusion of the VSSF-CTW. The VSSF-CTW treated the liquid digestate, considered loss for case 0, provided conditions for plant growth, and recovered sediments. Overall, the SPAD plus the VSSF-CTW was not considered a sustainable system from an exergetic perspective, since the balance between inputs and outputs was negative. Additionally, the SPAD with VSSF-CTW was not considered sustainable during the rainy or dry period (Table 6.5). The ability of the VSSF-CTW to store and export water in the rainy period (45,953 ± 3,860 kg/week) was significantly higher than in the dry period (12,371 ± 2,510 kg/week; $p < 0.001$), based on the Mann-Whitney test for two independent samples. However, the higher output of water (2,297 ± 193 MJ/week) was partly cancelled out by water inputs by precipitation and runoff (1995 ± 656 MJ/week) from August to December 2015, and the net exergy output of stored water in the VSSF-CTW was higher during the dry period (618.6 ± 125 MJ/week) than during the rainy period (302.3 ± 849.4

MJ/week). Thus, more exergetically favorable $\eta_{env,ex}$ and $R_{pol,ex}$ values during the dry months were obtained, in which the VSSF-CTW only stored treated liquid digested (Table 6.5).

In case 1, labor consisted of 2.5 hours per week to operate equipment for case 0 and an additional two hours for case 1. These two hours per week consisted of weekly activities such as overseeing discharge of liquid digestate effluent into the VSSF-CTW and harvesting plant biomass. The geotextile was cleaned of sediments only three times during the period of this study, which represented as 30 minutes of the two hours per week. Considering labor (353 MJ/week) in case 1, the $\eta_{env,ex}$ and the $R_{pol,ex}$ would be 0.684 ± 0.179 and 0.462 ± 0.383, respectively (Table 6.4 and 6.5). Labor impacted sustainable indexes negatively by a reduction of 6.80% of $\eta_{env,ex}$ and an increase of 27.5% of $R_{pol,ex}$. In fact, 157 MJ/week were employed for harvesting plants, while harvested biomass produced a net output of only 4.13 MJ/week. In contrast, sediments collected at the geotextile produced an exergy of 69.2 MJ/week after considering the 39.3 MJ/week needed for cleaning the geotextile. It is important to note that determining the effect of plant harvesting or the geotextile on treatment by VSSF-CTW was beyond the scope of this chapter; consequently, the above exergy comparisons for biomass harvesting and geotextile use do not reflect changes in the exergy of the VSSF-CTW effluent that may results from these practices.

6.3.3. Case 2 – SPAD-HCTW

The total input exergy for case 2 was 5,974 ± 1,239 MJ/week (Table 6.4). Case 2 included the FWS-CTW to further treat effluent from the VSSF-CTW. Additional inputs to case 2 corresponded to the electricity to run the FWS-CTW recirculation pump, and precipitation and runoff (Table 6.2). This pump only represented 0.371% of the total exergy inputs in case 2.

Similarly to case 1, optimization of operation of the FWS-CTW recirculation pump would not yield a substantial reduction in the exergy inputs. Similarly, input exergy would decrease to 4,295 ± 582 MJ/week if the wetland were completely isolated from runoff.

Total output exergy for case 2 was 33,458 ± 2,389 MJ/week (Table 6.4). Outputs from case 1 corresponded to 10.6% of the exergy output, while 13.4% was provided by treated water from the FWS-CTW. Approximately 2.82% of exergy output resulted from harvesting of plant biomass and 73.1% corresponded to non-harvested plants accumulated at the FWS-CTW.

From August to December 2015, water in the FWS-CTW consisted of effluent from the VSSF-CTW and precipitation and runoff. During that rainy period, all effluent water quality parameters except TS met discharge limits for Costa Rica as final effluent concentrations were 243 ± 23 mg TS/L, 34.3 ± 2.0 mg COD/L, 2.03 ± 0.45 mg TN/L, and 0.913 ± 0.145 mg TP/L. Similarly to the VSSF-CTW, the effluent from the FWS-CTW was considered an output because the water can be used for any purpose (e.g., irrigation or reuse at the SPAD). The mineral origin of the TS (low COD concentration and the low VS to TS ratio (34.7 ± 5.1%)), the fact that the FS concentration (158.7 mg/L) was below to the 200 mg TS/L allowed for surface discharge from livestock activities (MINAE-MSP 2007), and the comparison of TS concentration of liquid manure applied to the land, justify the decision to consider the effluent from the FWS-CTW as reclaimed. Reclaimed water from the FWS-CTW allowed the use of the effluent for irrigation and reuse at the SPAD in the rainy period (16 weeks). This was 2,017 ± 828 MJ/week and 263.7 ± 15.0 MJ/week for irrigation and reuse at the SPAD, respectively, out of the net exergy of water stored during the rainy period (6,514 ± 618 MJ/week).

In contrast, from January to March 2016, the FWS-CTW only received treated effluent from the VSSF-CTW and final effluent concentrations of 892 ± 69 mg TS/L, 288 ± 57 mg

COD/L, 15.7 ± 6.8 mg TN/L, and 0.632 ± 0.084 mg TP/L indicated that only TN and TP met discharge standards in Costa Rica. However, as the FWS-CTW effluent was used for operating the SPAD during the dry period, this water was valuable and considered as an output in the exergy balance. In the dry period, 208 ± 28 MJ/week were used from the net exergy of water stored at the FWS-CTW (1,767 ± 456 MJ/week).

As the FWS-CTW effluent can be used for any purpose during the rainy months (e.g.., irrigation or reuse at the SPAD) or during the dry period (e.g.., reuse at the SPAD), the exergy value was estimated by considering the standard chemical exergy of the water and totaled to 4,480 ± 601 MJ/week (Table 6.4). Similarly, for water with low concentrations of COD, TN, and TP, the estimation of the exergy based on COD, TN, and TP standard chemical exergies would be low, 61.8 ± 8.17 MJ/week, and the net exergy output due to the treated water at the FWS-CTW would increase to 4,541 ± 609 MJ/week.

Biomass harvested at the FWS-CTW corresponded to 2.82% of the exergy outputs in case 2. The exergy output from harvesting of floating plants depended plant densities at the time when plants were harvested. These aquatic plants, especially the *Eichhornia crassipes*, spread rapidly across the surface in the water, and the accumulated biomass stored at the FWS-CTW totaled to 24,473 ± 1,362 MJ/week, which was the 73.1% of the exergy outputs in case 2. In this study, reported growth rates for *Eichhornia crassipes* and *Pistia stratiotes* were 5.25 ± 0.28 and 1.12 ± 0.05 kg DM/m^{2}week, which were 12.5 to 37.5 and 2.7 to 8.00 times higher than the growth rates reported by Gutierrez, Ruiz et al. (2001) and Gupta, Roy et al. (2012) (from 0.14 to 0.42 kg/m^{2}week). If their maximum growth rate (0.42 kg/m^{2}week) for both *Eichhornia crassipes* and *Pistia stratiotes* were used, the exergy output due harvested biomass at the FWS-CTW would have decreased to 60.8 and 67.5 MJ/week for *Eichhornia crassipes* and *Pistia stratiotes*, respectively. The plant biomass stored at the FWS-CTW would decrease to 1,644 MJ/week for *Eichhornia crassipes* and 1,824 MJ/week for *Pistia stratiotes*. Considering this conservative growth rate, the total output exergy for case 2 would decrease to 11,637MJ/week. However, the balance between exergy inputs and outputs would remain positive. In terms of plant growth rate, the FWS-CTW produced 228 ± 38 times more exergy than the VSSF-CTW. Reported growth rates by Opio, Jones et al. (2014) and Zhang, Rengel et al. (2007) for *Cyperus papyrus* (16.7 to 37.4 g/m^{2}d) and *Canna indica* (33.5 ± 0.3 g/m^{2}d) shows that biomass production is lower by 11 times the growth rates reported by Gutierrez, Ruiz et al. (2001) and Gupta, Roy et al. (2012) (from 0.14 to 0.42 kg/m^{2}week) for plant type, which support the observation that FWS-CTW can obtain higher exergy output from plant biomass than VSSF-CTW systems.

For case 2, the $\eta_{env,ex}$ was 5.60 ± 1.56 and the $R_{pol,ex}$ was -0.821 ± 0.167 (Table 6.5). Sustainable indexes were higher than 1 and lower than 0 for the $\eta_{env,ex}$ and the $R_{pol,ex}$, respectively. This study was conducted to answer the question *is the SPAD-HCTW exergetically sustainable?* Thus, for the period of this study, the SPAD-HCTW was sustainable from the exergetic point of view. The exergy outputs increased by 1,800% and 800% with respect to case 0 and case 1, respectively, because of the capacity of the HCTW to treat and store water and produce biomass (harvested and stored plants).

Addition of the FWS-CTW to the system positively impacted the exergy balance between inputs and outputs, mainly because the greater storage capacity of the FWS-CTW allowed for storage of runoff and precipitation that would otherwise be considered a loss of exergy in the system to be converted into useful water (an exergy output). Because the filter media (32% of porosity) occupies storage volume in the VSSF-CTW, the FWS-CTW stored more water, thus higher output exergy was achieved with the FWS-CTW in terms of water storage useful for any purpose (e.g., irrigation or reuse at the SPAD). The FWS-CTW processed 130,287 ± 12,367 kg/week of water during the rainy months, in contrast to 35,351 ± 9,136 kg/week of water during the dry months, volume of water that were 2.84 and 2.14 times the volume stored at the VSSF-CTW. The exergy output of the stored water was significantly higher in the rainy period (6,514 ± 618 MJ/week) than in the dry period (1,325 ± 456 MJ/week), with a p value <0.001. However, the dry period obtained better $\eta_{env,ex}$ and $R_{pol,ex}$ values than during the rainy for case 2 (Table 6.5). Input during the rainy period was higher than during the dry period due to the input of rain and precipitation (1,995 MJ/week versus zero in the dry period), but with respect to the input, higher output was obtained during the dry period (input=3,124 MJ/week, output=29,237 MJ/week) than during the rainy period (input=7,755 MJ/week, output=36,074 MJ/week) due to the constant growth of plants in both periods.

Labor at the FWS-CTW was one hour per week to harvest plants. In addition to the four and half hours per week from case 0 and case 1, the total exergy by labor for the case 2 was 432 MJ/week. Labor decreased the $\eta_{env,ex}$ by 6.74% to 5.22 ± 1.38 and increased the $R_{pol,ex}$ by 1.57% to -0.809 ± 0.166 with respect to the case 2 values. Even when exergy due to labor was considered in case 2, the SPAD-HCTW was considered borderline sustainable, in contrast to case 0 and 1. In fact, the exergy for harvesting the plants was only 8.34% of the exergy gained by the harvesting of plants (943 ± 49 MJ/week).

Table 6.5 includes the capital expenditure (CaPex) and net energy produced for the three cases (Aguilar Alvarez, Bustamante Roman et al. 2016). Addition of the HCTW to the SPAD increased costs by 23.0% and 35.0% for case 1 and case 2, respectively, while the net energy decreased by 1.24% and 2.42% for case 1 and 2, both with respect to case 0. These two negative impacts in costs and energy consumption can be offset by potential sustainability of the SPAD-

HCTW. The SPAD-HCTW had net revenue of $2,436 per year and net energy of 1,476 MJ/week (Aguilar Alvarez, Bustamante Roman et al. 2016). However, this basic energy and expenditure analysis did not provide insight into the environmental benefits that were observed from inclusion of the HCTW as a treatment system for the SPAD digestate. By this exergy assessment of sustainability, it was seen that energy production, reuse of agricultural residues, and water treatment contributed to environmental benefits accrued by the SPAD-HCTW.

6.4. Conclusions

Sustainability is a complex and broad concept, which varies in interpretation depending on the discipline and context. The United Nations define sustainable as *"the process that constrains resource consumption and waste generation to an acceptable level, makes a positive contribution to the satisfaction of human needs, and provides enduring economic value to the business enterprise"*. The SPAD-HCTW constrained resource consumption and waste generation while producing energy, fertilizer, and treated water. These outputs represent positive contributions to the satisfaction of the human needs. The exergy-based assessment of sustainability from two indexes ($\eta_{\mathrm{env,ex}}$ and $R_{pol,ex}$) indicated net environmental benefits from the SPAD-HCTW from a thermodynamic perspective. Future work is needed to quantify additional economic, social, and environmental aspects of sustainability with regards to the SPAD-HCTW.

From an exergy perspective, the baseline SPAD (case 0) was not sustainable, based on $\eta_{\mathrm{env,ex}}$ and $R_{pol,ex}$. By including the HCTW as treatment for digestate from the SPAD, the system became thermodynamically sustainable, not only producing energy and fertilizer, but also treated water - an output that was not achievable with the SPAD alone. Due to a positive exergetic balance (inputs < outputs), $\eta_{\mathrm{env,ex}}$ and $R_{pol,ex}$ had values higher than 1 and lower than 0, respectively. This result was achieved by conversion of biomass residues and agricultural wastewater (e.g., food waste, chicken litter, and treated water) into high quality end products (energy, fertilizer, and treated water) when combining solar technology with anaerobic digester and HCTW treatment.

For the entire SPAD-HCTW, energy, treated water, and plant biomass were the main outputs from the conversion of raw material into high quality end products. The SPAD (case 0) produced energy from current problematic wastes in Costa Rica and can be offered as an option for alleviating waste problems and energy demands. As a post-treatment process to treat water and produce plant biomass, the HCTW treated the liquid digestate and avoided potential surface and ground water contamination by direct discharge to land or surface water. In addition, the HCTW, especially the FWS-CTW, became a source of water which had higher storage capacity than the VSSF-CTW. This storage capacity turned in higher exergy output by water as the FWS-CTW had higher holding capacity during the rainy period. Interestingly, the plants at the FWS-CTW showed higher growth rate than the plants at the VSSF-CTW, with a consequently higher exergy output by plant biomass.

The exergy-based assessment of sustainability was not impacted by the precipitation. The SPAD alone did not show differences between the rainy and dry period as no elements of the SPAD were affected by rain. However, precipitation and runoff were included as inputs in case 1 and 2. Positively, the VSSF-CTW and FWS-CTW had enough storage capacity to hold water from precipitation and runoff. However, no differences between the rainy and dry months were found as the exergy output of water stored in both the VSSF-CTW and the FWS-CTW were partly cancelled out by water inputs by precipitation and runoff. In general, more exergetically favorable $\eta_{env,ex}$ and $R_{pol,ex}$ values during the dry months were obtained due to better digestion performance (case 0), storage of water for future use, and biomass production (harvested and non-harvested plants) (case 1 and 2).

Biomass production marked a difference in the exergy balance, because independently of the period, the growth rate was constant and, in the particular case of *Eichhornia crassipes*, fast. Plants converted the nutrients in the wastewater into an organized structure (steam, leaves, flowers, etc) and provide habitat, which in turn increase biodiversity (Jørgensen 2006, Tang, Fang et al. 2014). Biomass harvested was valuable as a source of nutrients after composting; however, the exergy output from nutrients was not remarkable. Other processes could be explored for utilizing the biomass, especially the *Eichhornia crassipes*, which had a fast growth rate. *Eichhornia crassipes* could be used as a substrate for anaerobic digestion as long as a pretreatment process removes the lignin (Bharati and Kalamdhad 2016). Recent processes such as electrohydrolysis pretreatment (Barua, Raju et al. 2017), thermal pretreatment (Barua and Kalamdhad 2017), microwave pretreatment (Budiyono, Sumardiono et al. 2015), and older process such as chopping and drying (Moorhead and Nordstedt 1993), could be studied for creating another the loop in the SPAD-HCTW. For example, Budiyono, Sumardiono et al. (2015) estimated that the microwave pretreatment of *Eichhornia crassipes* at 560 W for seven minutes double the biogas production to 75.12 mL biogas/g TS from fresh when compare to the non-pretreated *Eichhornia crassipes* (37.56 mL biogas/g TS).

This study is the base for starting a protocol for determining sustainability of similar solid and liquid wastes management in exergetic terms. It was identified that for determining the exergy of the wastewater with high concentrations of contaminants (COD, TN, and TP) the approach developed by Tai and Matsushige (1986) and applied by Hellstrom (1997) and Martínez, Uche et al. (2010) is easier to apply than the calculation by the chemical composition of the wastewater. It is simpler to calculate the COD, TN, and TP concentrations in the wastewater and use the standard chemical exergy of COD, TN, and TP than to determine the elemental composition. For effluents with low concentration of COD, TN, and TP, the use of the COD, TN, and TP standard chemical exergy will be low. For these cases, the exergy of the water can be calculated using the standard chemical exergy of the water, and depending the purpose of the treated water (e.g., irrigation, reuse at the SPAD), could include the exergy value due to the COD, TN, and TP concentration.

In conclusion, the SPAD-HCTW (case 2) was exergetically sustainable. Some economic, social, and environmental benefits can be deduced from this study. Economically, energy, fertilizer, and water reclamation could represent savings for families at rural areas. Socially, savings could cover other important needs (e.g.: education, health, etc.) and the SPAD-HCTW became another solution to provide water treatment. Environmentally, treatment of biomass residues and agricultural wastewater could avoid the emission of greenhouse gases, as well as the impairment groundwater and surface water. However, to better understand the sustainability benefits of the SPAD-HCTW, more analyses with additional economic, social, and environmental metrics are needed.

6.5. Acknowledgment

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CHAPTER 7: CONCLUSIONS

7.1. Biomass residues and wastewater: the problem becoming an opportunity for sustainable development

Costa Rica is striving to be the first neutral carbon country for 2021. One front to carbon neutrality is generation of electric energy using renewable resources, replacing fossil fuel based electricity. Excluding transportation, which depends 100% on fossil fuels, Costa Rica generates mainly electric energy from hydric (75%), geothermic (12%), wind (10%), solar (0.01%), and biomass (0.73%) resources (ICE 2016). Only 1.8% of electric energy is produced from fossil fuels. Even though the installed power thermic capacity generation in Costa Rica is 200 MW (8% of the total power capacity in the country), the country only uses this thermic plant as a backup (Leandro 2011). Indeed, in 2016, the country produced electric energy from renewable resources for 271 days in a row (Raedle 2017). Precipitations and rivers fed reservoirs that allowed a continuous generation of renewable electric energy, even when El Niño-Southern Oscillation (ENSO) impacted the country with less precipitation (IMN 2016). This is a warning sign; the reliance of Costa Rica on hydroelectric energy can turn negative as climatic phenomena (e.g., ENSO) can bring severe droughts. For example, in 2013, the water level of Arenal Lake, the main artificial pluriannual reservoir for hydric power generation in Costa Rica, almost depleted to a non-productive point (Aguero 2013). Thus, to prevent fossil utilization for electric energy production, other renewable electric energy generation approaches need to be adopted and strengthened. For example, Coto (2013) demonstrated that agricultural organic residues have a potential electric energy generation of 600 MW; this would be 23% of the total power capacity in the country.

In addition to its carbon neutrality goal, Costa Rica is marketing itself as a green country, committed to nature. This is an economic strategy for attracting investors to green development, an essential for Costa Rica. One front is the protection of the environment (e.g. national parks and private refuges) and the electric energy generation from renewable resources.

In addition, Costa Rica promotes the optimization of resource utilization (e.g., minimum utilization, reutilization, and recycle) in any economic activity. For example, business should use less energy, water, and materials in production and operational lines. These efforts are awarded by the Ecological Blue Flag. However, the majority of the businesses follow a linear economy model (make, use, dispose), in which there is awareness in the use of resources, but no awareness of the management of sub-products (e.g., solid and liquid wastes). In fact, as stated by Calvo (2014), "Costa Rica has invested greater efforts to conserve bio-diversity than to pollution control and waste disposal". Consequently, the environment, in particular surface and groundwater resources, is impaired, and contradictorily, this "green" country has a severe sanitation and water pollution problem.

Moreover, there is a global challenge. More food, energy, and water are demanded by a growing population, while current environmental degradation and climate change effects restrict the access to these resources (FAO 2014). The food-energy-water nexus is a conceptual tool for balancing the natural resource supply and demand for achieving sustainable development (FAO 2014, Biggs, Bruce et al. 2015). Once the linkage between these three resources is recognized in a particular economy, integrated systems can be implemented to optimize resource supply and consumption. The key is to create close-looped systems to convert linear into circular economies (Agrocycle 2017). In contrast to a linear economy, a circular economy considers residues as resources. In turn, circular economy alleviates the demand of resources and reduce waste disposal. Figure 7.1 depicts how integrating a treatment system, the linear model can be converted into a circular one. The treatment system processes the residues to convert them into to resources, decreasing resource demand from the environment and decreasing waste disposal into the environment.

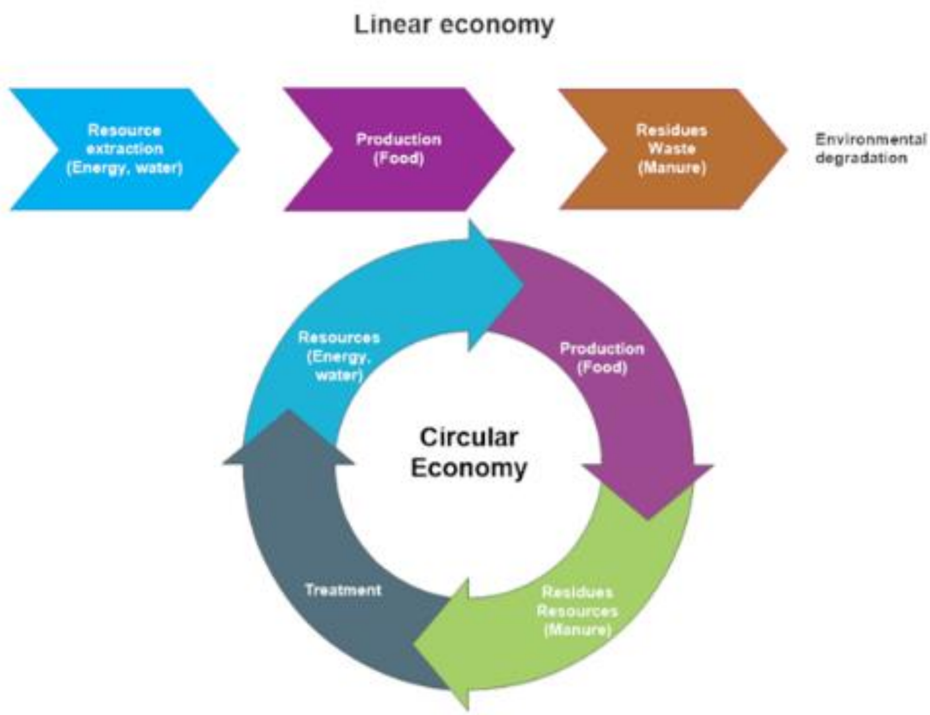

Figure 7. 1. Linear versus circular economy (Own creation).

Even with Costa Rican efforts toward carbon neutrality and green development, there are gaps have not been considered, while environmental degradation and climate change are restricting even more resources. Thus, there is an opportunity to transform current linear economies into circular ones by the integration of innovative treatment systems. Innovative treatment systems can partially solve the water pollution problem at Costa Rica and produce energy from biomass resources, while protecting the environment and decreasing the demand from the current electric grid. This clearly will strengthen Costa Rican green development and the progress towards carbon neutrality. In particular, biosystems engineering can focus on agro-industry and agricultural activities in rural areas of Costa Rica. The idea is to provide a decentralized self-sufficient, close-looped, organic waste treatment system technology as a solution to linear development in agro-industry and agricultural activities located in rural areas of Costa Rica.

The present document proposed one approach for solving the problem of water pollution due to agro-industrial and agricultural waste disposal in rural areas of Costa Rica. The proposed integrated system consisted of three well-known technologies: solar thermal collection, anaerobic digestion, and constructed treatment wetland. All together created a decentralized self-sufficient, close-loop, organic waste treatment system, which in turn, provided energy, fertilizers, and treated water. The solar-powered anaerobic digestion and hybrid constructed treatment wetland (SPAD-HCTW) utilized the residues from current linear production systems to convert them into resources and create a circular, close-loop, system. The present dissertation demonstrated the technical performance and sustainability of the SPAD-HCTW in Costa Rica (Figure 7.2).

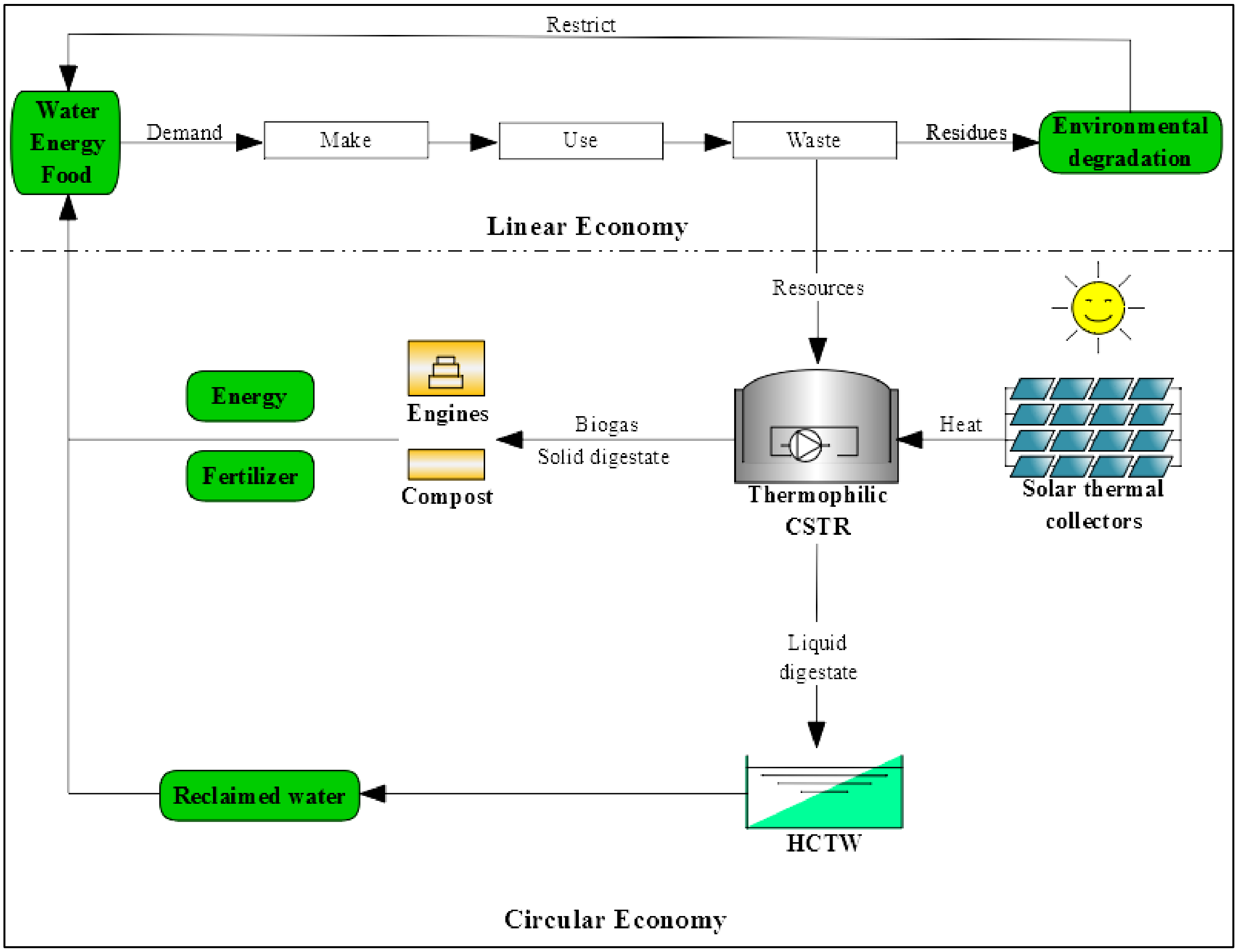

Figure 7. 2. Closing the loop for water, energy, and food (Own creation).

Technical performance of a solar-powered waste utilization and treatment system in Costa Rica, chapter 3, demonstrated that the solar thermal collector unit provided sufficient energy to maintain thermophilic temperature in a 20 m^3 CSTR anaerobic digester. Under thermophilic conditions, the CSTR anaerobic digester converted organic wastes (e.g., food waste and chicken litter) into energy and fertilizers. The entire treatment system was self-sufficient and surplus energy was obtained. In terms of treatment of wastewater, it is well-known that under thermophilic conditions pathogens are killed; however, the liquid digestate still had high concentrations of organic matter, solids, and nutrients. Then, to assure wastewater treatment, the vertical flow subsurface (and free water surface) constructed treatment wetland treated the digestate. Treated water was reutilized either for irrigation or for diluting the feedstock. This final step assured the recovery of resources (energy, fertilizers, and water) and converted a linear model into a circular, close-loop, system. In general, 263 MJ renewable energy, 28 kg nitrogen and phosphorus fertilizer, and 550 kg of treated water were generated from 863 kg of mixed animal and food wastes. The net revenue considering electricity and fertilizer was $2,146 annually. The payback period for the system was estimated to be 21 years; however, a sensitivity analysis demonstrated that through optimization, the payback period can be reduced to 9 years.

The proposed system is innovative compared to decentralized systems developed at farms in Costa Rica. In Costa Rica, the Costa Rican Electricity Institute (ICE) leads the installation of anaerobic digesters. ICE promotes anaerobic digestion for producing energy from manure (cow, swine, and chicken litter), while treatment of wastewater is not a priority. In general, anaerobic digesters are tubular, plug-flow bags (e.g., tubular polyethylene bag digesters) for small-scale farms, usually 10 to 15 heads. In addition, covered lagoon anaerobic digesters are used for larger farms (e.g., Noble farm: 120 cows; and Kafur: 4,000 swine). These systems are meant to be mesophilic (e.g., 35°C); however, mesophilic temperature is rarely reached as the systems depend on ambient temperatures (Kinyua 2015). Even though the production of biogas is achieved, the digestion process is neither efficient reducing solids, organic matter nor for killing pathogens. Then, digestate is not post-treated and it is simple discharged. Usual practice is land application of the liquid digestate; however, there is a potential health impact due to the transmission of pathogens to food and water, as well as an impairment to ground and surface water due to high concentration of nutrients (Kinyua 2015).

Performance of the hybrid constructed treatment wetland treating digestate effluent in Costa Rica, chapter 4, demonstrated that the HCTW had sufficient water storage capacity and achieved a treatment performance that allowed the reuse of the treated water in other activities (e.g., irrigation and reuse at the SPAD). The hybrid configuration was important to overcome the disadvantages each individual wetland had. The VSSF-CTW received high strength liquid digestate and demonstrated high reduction of solid and organic matter, as well as high transformation of ammonium to nitrate. Then, the FWS-CTW served as a polishing step for the effluent from the VSSF-CTW, in particular, in the transformation and removal of nitrate through denitrification. Finally, the study of natural treatment systems, such as the HCTW, require larger campaign evaluation (years) for establishing central tendencies in the treatment. Thus, during this evaluation (from August 2015 to March 2016), it was not possible to represent a narrow central tendency using loading charts or contaminant removal models, given the variability in the digestate and environmental conditions.

The HCTW is innovative compared to constructed treatment wetlands installed in Costa Rica. To our knowledge, this is the first HCTW treating high strength wastewater from agricultural residues in Costa Rica. Few wetlands have been installed in Costa Rica. These systems were built to treat grey and black water (Dallas, Scheffe et al. 2004, Alfaro, Perez et al. 2013, Pérez, Alfaro et al. 2013). For rural areas, this dissertation, as well as studies by Dallas, Scheffe et al. (2004), Pérez, Alfaro et al. (2013), and Alfaro, Perez et al. (2013), demonstrated that the sanitation and water pollution problem at Costa Rica can be overcome by implementing decentralized wastewater treatment systems such as CTWs. In fact, the implementation of decentralized wastewater treatment systems is recommended by Naik (2014). Naik (2014) indicated that the high capital investment (e.g., collection system and the treatment plant) makes the implementation of centralized system in rural regions unfeasible. In addition, centralized treatment systems have shown to be unmanageable in some places due to technically skilled manpower for operation and maintenance (Dallas, Scheffe et al. 2004, Avila, García et al. 2016). For example, the Costa Rican Water and Sewage Institute (AyA) reported in 2003 that centralized wastewater treatment plants at the cities of Heredia, Alajuela, and Cartago, are not operating, and wastewater is instead directly discharged into rivers (Araya, Araya et al. 2003). The situation has not changed at all in recent years, and sewage collectors in those cities simple discharge the wastewater into the rivers (Calvo 2014).

Infiltration measurements, chapter 5, demonstrated that after three years of continues operation, the VFSS-CTW was not clogged. Even with the high strength characteristics of the liquid digestate, strategies implemented to prevent clogging have positively impacted in the longevity of the filter media. The geotextile membrane removed solids, allowed the recovery of nutrients (solids retained by the membrane), and avoided the direct contact of solids with the filter media. The implementation of a geotextile membrane in the inlet area provided a low-cost, low-maintenance approach to improving the performance of the VSSF-CTW. The effective treatment surface area only received treated water through the recirculation system. Under this condition, the estimated void space at the filter media was 20 of 30 m^3. At the field scale, no differences in volatile solid accumulation, root development, and infiltration rates in the filter media were found based on plant type. However, in laboratory-scale columns, columns planted with *Canna* and *Cyperus* exhibited different infiltration rates, despite similar volatile solid accumulation and root development, indicating that root morphology of *Canna* could be favorable to infiltration.

Exergy-based assessment of sustainability of a solar-powered anaerobic digestion and hybrid constructed treatment wetland system to treat agricultural wastes in Costa Rica, chapter 6, demonstrated that SPAD-HCTW was a tool for sustainable development. Sustainability is a broad concept, and herein was limited to a system that constrains resource consumption and waste generation to an acceptable level, makes a positive contribution to the satisfaction of human needs, and provides enduring economic value to the business enterprise (UN 2015). The SPAD-HCTW processed residues (e.g.: wastes) and converted then into resources (e.g.: energy, fertilizers, and treated water). Thus, the integration of the SPAD-HCTW as a decentralized self-sufficient, close-looped, organic waste treatment system into agro-industry and agricultural activities located in rural areas of Costa Rica would:

- Constrain waste generation. The SPAD-HCTW treats waste from agro-industrial and agricultural wastes.
- Constrain resource consumption. Treating wastes, the SPAD-HCTW generates energy, fertilizers, and water, that can be reused.
- Satisfy human needs. The SPAD-HCTW provides wastewater treatment, which in turn brings environmental and public health benefits. Environmentally, the electric generation from biomass can increase, reducing the need of fossils and strengthening the neutral carbon process to 2021. In addition, the SPAD-HCTW alleviates the severe water pollution problem at Costa Rica.
- Economic value. Saving from energy generation and fertilizers. Per day, the SPAD-HCTW generated 263 MJ renewable energy from 863 kg of mixed animal and food wastes, which produced net revenue of $2,146 annually from electricity and fertilizer.

From the technical point of view, the SPAD-HCTW was a sustainable technology that produced energy, fertilizers, and treated water. Integration of technologies was key to overcome the disadvantages of the individual technologies. The higher energy requirement of thermophilic anaerobic digestion can be supplied by solar thermal collectors. By itself, the solar thermal collector represents an unsteady energy flow for solar power generation. Instead, the thermophilic anaerobic digestion converts low-density and inconsistent solar energy (as heat) into a relatively dense and reliable biochemical energy source – methane. Together, anaerobic digestion and solar thermal collection technology provide energy, fertilizers, and reduce greenhouse gas emissions; however, water reclamation is not possible. Thus, a post-treatment system is needed for wastewater treatment. The HCTW demonstrated to the capability of treating strength wastewater. The treatment performance depends on the wetland status, and preventive strategies prevented clogging, one of the commonest problems of wetlands.

7.2. Future work

The SPAD-HCTW can become one approach to solving Costa Rica's water pollution problem, while advancing Costa Rica toward carbon neutrality and the green development. However, there are still aspects to overcome in the future related to this technology.

There is a need to optimize the SPAD-HCTW for reducing costs to make this approach more economically attractive. It is clear that the SPAD-HCT provides environmental benefits; however, the large payback period is not attractive. Future research should focus on modeling and validation of the solar thermal collection unit and on minimizing range of temperature variation inside the digester. Modeling and validation can be based on energy demand by the anaerobic digester and average annual environmental temperature and irradiance at the site for optimizing solar thermal collector area (Suryawanshi, Chaudhari et al. 2010). In the tropics, this will be a useful tool for designing solar-powered digesters based on the organic waste and region where the system will be installed. For example, in China, Hassanein, Qiu et al. (2015) modeled and validated a solar greenhouse to surround the anaerobic digester with the intention to maintain minimum temperature for anaerobic digestion. The validated model allowed determination of geographic zones to house the proposed solar greenhouse unit in China. Then, controls can be implemented (e.g., PIDs) for promptly turning on and off heat transfer pumps to avoid large temperature variation inside the digester (Alkhamis, El-khazali et al. 2000).

Moreover, the experience gained in the operation of the SPAD narrowed topics for future studies. For example, more studies are needed to evaluate parameters such as mixing frequency and intensity (Zabranska, Dohanyos et al. 2002, Bombardieri, Espinosa-Solares et al. 2007, Suryawanshi, Chaudhari et al. 2010), feed frequencies (Bombardieri, Espinosa-Solares et al. 2007), and TS content in the feed (Espinosa-Solares, Valle-Guadarrama et al. 2009).

More data collection at the HCTW is needed to narrow central tendencies in treatment performance of the wetlands. In addition, the geotextile membrane implemented in the VSSF-CTW was important, and more studies are needed to characterize its impact on treatment performance of the VSSF-CTW. Some questions for future research are:

- Did the geotextile create short circuiting and dead points within the VSSF-CTW? Treatment of the primary influent was assumed to occur predominantly beneath the geotextile, leaving most of the treatment area for treatment of the recirculated wastewater. However, the mass balance indicated that diffusion of solids beneath the geotextile likely occurred.

- Was there any relationship between sediment accumulation in the geotextile and the effluent concentration from the VSSF-CTW? As the sediments accumulated in the geotextile, more sediments were filtered out, likely improving sediment removal.
- How much could the main treatment area be decreased due to the geotextile?

In addition, the geotextile membrane could prevent clogging of the sand media of the VSSF-CTW. From this observation, a principal question is proposed for further research and optimization of the vertical-flow wetland: *How do the solids distribute as the wastewater moves downward through the filter media below the geotextile membrane?* A tracer study is proposed for determining dynamics of the solids across that portion of the wetland. Distribution, retention of solids, and time to clogging can be impacted by precipitation and flooding events and the real condition of the filter media beneath the geotextile membrane is currently unknown. In addition, there is a need to identify morphological traits of root plants, particularly *Canna* plants which did not adversely affect infiltration in the full-scale wetland.

Finally, environmental benefits can justify special incentives by the government to promote this technology in the country. As depicts in Figure 7.3, this dissertation has shown the environmental benefits from the SPAD-HCTW; however, an extended sustainable study considering the SPAD-HCTW as part of the surrounding ecosystem and community needs to be conducted to elucidate how the decentralized self-sufficient, close-looped, organic waste treatment system also brings economic and social benefits. Holistic approaches such as the SPAD-HCTW can partly alleviate the water pollution problem; however, as concluded by Calvo (2014) the country needs to politically prioritize sanitation and water pollution problems. Until that happens, academic researchers need to promote and encourage the target population to implement this approach by using the best available weapon in Costa Rica: environmental education.

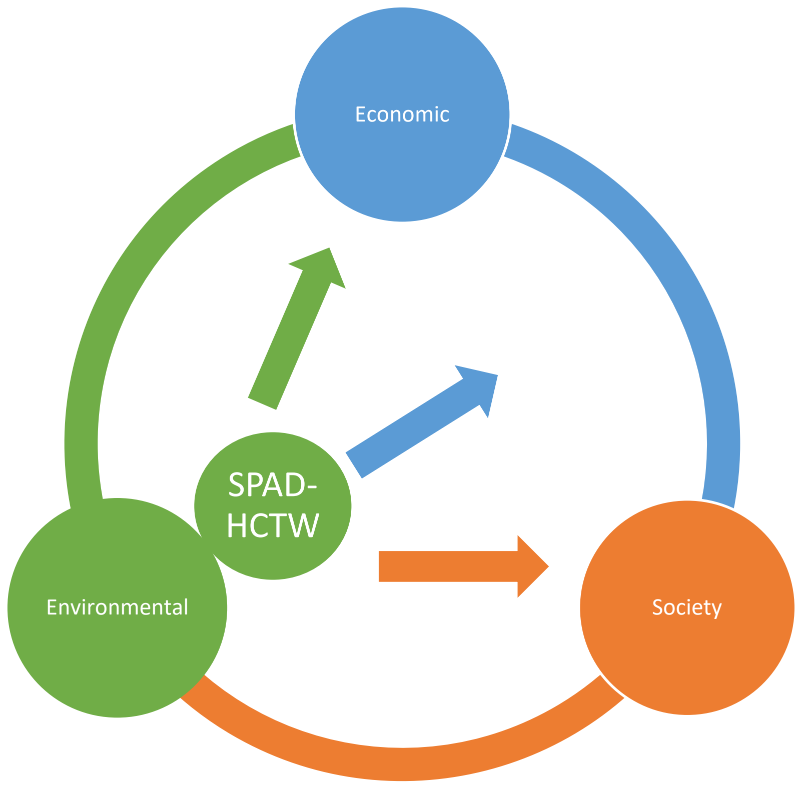

Figure 7. 3 . The SPAD-HCTW as a sustainable approach for the sanitation problem in Costa Rica.

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