Removal and Recovery of Nutrients from Wastewater in Urban and Rural Contexts

by

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DEDICATION

I dedicate this dissertation to my wife, Sarah, for her daily love, encouragement, sacrifice, and ability to make each day a beautiful adventure; to my parents, Keith and Brenda, for raising me to care for others, and supporting my trailblazing even when they felt the adventures were a little bit crazy; to my co-advisors, Dr. Jeffrey Cunningham and Dr. James Mihelcic, for going above and beyond to guide my academic path; and to the memory of Dr. Peter Bosscher, for modeling how to live a life that integrated personal values, engineering skills, and real-world challenges.

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ABSTRACT

Efforts to remove and recover nutrients from wastewater are motivated by the United Nations Sustainable Development Goals and the National Academy of Engineering Grand Challenges of Engineering. Of the seventeen Sustainable Development Goals (SDGs), multiple SDGs relate to managing nutrients in wastewater. SDG 6, which is to "ensure availability and sustainable management of water and sanitation for all," contains targets that aim to improve water quality by reducing pollution, halve the amount of untreated wastewater released to the environment, and increase recycling and safe reuse of wastewater (UN, 2017). SDG 2 seeks to improve food security and SDG 12 seeks to sustainably manage natural resources. Similarly, the National Academy of Engineering Grand Challenges of Engineering highlight managing the nitrogen cycle and providing access to clean water (NAE, 2019).

Centralized wastewater treatment plants (WWTPs) have historically been designed to remove nutrients (such as nitrogen and phosphorus) and other contaminants prior to discharge. Modern wastewater treatment practices integrate recovery of resources including nutrients, energy, and water. The many available technologies, coupled with competing priorities, can complicate community decision-making on the choice of technology and the scale at which to implement the technology (i.e. building, community, or city), as well as determining how new upstream treatment may affect existing downstream treatment. Technologies that recover energy or manage nutrients such as anaerobic digestion, struvite precipitation, and microbial fuel cells can be implemented at a variety of scales in urban settings and may also be viable for influent

types such as agricultural waste. Therefore, the overall goal of this dissertation is to contribute to the achievement of multiple sustainable development goals through the removal and recovery of nitrogen and phosphorus from a variety of influents at a variety of scales.

One type of decision-making tool that assists in the choice of nutrient management technologies is a House of Quality. I developed a tool based on the House of Quality that integrated multiple priorities at three scales in a sewershed and produced rankings that generally align with current wastewater treatment practice. Accordingly, top-ranked city-scale technologies are those commonly employed (e.g. A^2O, oxidation ditch) that use the dissolved organic carbon present in the wastewater to drive denitrification. Similarly, conventional treatment (e.g. flush toilet connected to a sewer) is ranked highest at the building scale because of its easy maintenance, small footprint, and inoffensive aesthetics. However, future trends such as technology development will likely affect the technologies, weightings, and scores and therefore improve the ranking of novel and emerging technologies. This trend may be amplified by the implementation of test beds, which can provide opportunities to improve the technical characteristics of developing technologies while minimizing risk for municipalities.

The House of Quality planning tool was utilized in an in silico case study to analyze nutrient management technologies at three scales across the Northwest Regional Water Reclamation Facility sewershed in Hillsborough County, FL. The study demonstrated that employing treatment technologies upstream from the centralized wastewater treatment (i.e. building-scale source separation and community-scale technologies) could reduce nitrogen loading to the mainstream treatment train by over 50%. Sidestream treatment (i.e. the liquid effluent of anaerobic digestion that typically recycles back to the beginning of the mainstream treatment process) has minimal impact in nitrogen reduction, but is effective in reducing

phosphorus loading to the mainstream due to high quantities of phosphorus recycling back to the head of the plant. These results can inform decision-makers about which context-specific nutrient management technologies to consider at a variety of scales, and illustrate that sidestream technologies can be the most effective in reducing phosphorus loading while building- and community-scale technologies can be most effective in reducing nitrogen loading to the centralized treatment plant.

Struvite precipitation and microbial fuel cells (MFCs) can be used in combination to manage nutrients and recover energy in sidestreams of centralized WWTPs. Because the liquid effluent from engineered struvite precipitation often contains high concentrations of total nitrogen, I constructed and demonstrated a fixed-film nitrification reactor and a two-chambered MFC to further reduce total nitrogen and recover energy. The primary benefit of the MFC in the technology demonstrated here is not its ability to produce energy, but rather its ability to remove additional nitrogen through nitritation and denitritation. The sidestream nutrient removal prevents nutrients from returning to mainstream treatment, reducing operational costs. Such improvements to wastewater treatment processes can facilitate the transition to the resource recovery facility of the future by becoming a net-energy producer while also achieving the simultaneous benefits of nutrient recovery/removal and reduced costs associated with mainstream treatment.

Nutrients and energy can also be recovered in agricultural settings. In this dissertation I studied an agricultural waste treatment system comprising a small-scale tubular anaerobic digester integrated with a low-cost, locally produced struvite precipitation reactor. This study investigated two digesters that treated swine waste in rural Costa Rica. I also facilitated construction of a pilot-scale struvite precipitation reactor that was built on site using local labor

and local materials for approximately $920. Local products such as bittern (magnesium source) and soda ash (base) allowed for the production of struvite, a fertilizer that can replace synthetic fertilizer for rural farmers. Liquid-phase concentrations of PO_4^{3-}-P and NH_4^+-N in agricultural wastewater increased by averages of 131% and 116%, respectively, due to release from the swine waste during anaerobic digestion. Despite this increase in liquid-phase concentrations, an average of 25% of total phosphorus and 4% of total nitrogen was removed from the influent swine manure through sedimentation in the digesters. During struvite precipitation, an average of 79% of PO_4^{3-}-P and 12% of NH_4^+-N was removed from the waste stream and produced a solid with percentages (mass basis) of Mg, N, P of 9.9%, 2.4%, and 12.8%, respectively, indicating that struvite ($MgNH_4PO_4$) was likely formed. The treatment system offers multiple benefits to the local community: improved sanitation, removal of nutrients to prevent eutrophication, recovery of struvite as a fertilizer, and production of a final effluent stream that is suitable quality to be used in aquaculture. These are examples of how, more generally, quantifying nutrient recovery from agricultural waste and understanding recovery mechanisms can facilitate progress toward multiple sustainable development goals by improving sanitation, promoting sustainable management of wastes and natural resources, improving food security, and supporting local ecosystems.

Managing nutrients from a variety of influent types at different scales can contribute to the achievement of multiple sustainable development goals. Worldwide trends of population growth and resource depletion highlight the need for models to easily allow decision-makers the ability to understand the fate of nutrients and implement infrastructure accordingly.

CHAPTER 1: INTRODUCTION

1.1 Background and Motivation

Efforts to remove and recover nutrients from wastewater are motivated by the United Nations Sustainable Development Goals and the National Academy of Engineering Grand Challenges of Engineering. Of the seventeen Sustainable Development Goals (SDGs), multiple SDGs relate to managing nutrients in wastewater. SDG 6, which calls for universal access to water and sanitation, contains targets that aim to improve water quality by reducing untreated wastewater released to the environment and increasing safe reuse of wastewater (UN, 2017). SDG 2 (Zero Hunger) and SDG 12 (Responsible Consumption) also have targets that seek to improve food security through resilient and sustainable agriculture practices and reduce wastewater generation. Similarly, at the national level, the National Academy of Engineering Grand Challenges of Engineering highlight managing the nitrogen cycle and providing access to clean water (NAE, 2019). The Water Environment Federation also emphasizes the need to recover valuable resources like nutrients from wastewater (WEF, 2014).

Other global trends motivate the need for wastewater treatment to integrate recovery of resources such as nutrients (Guest et al. 2009; Orner et al. 2017; Mihelcic et al., 2017). First, the worldwide population is expected to exceed 9 billion by 2050, causing increased nutrient loading from urine and feces and increased demand for nutrient fertilizers. Approximately 22% of the global phosphorus demand could be met by collecting and harvesting the phosphorus found in human urine and feces (Mihelcic et al., 2011). Harvesting phosphorus from human urine and

feces is increasingly necessary as phosphorus reserves are unevenly distributed globally and could be depleted in the next 50-100 years (Cordell et al., 2009).

Nutrients can be managed in a variety of contexts—urban and rural, centralized and decentralized, developed and developing. In urban settings, a typical municipal wastewater treatment system consists of a centralized treatment plant that receives raw wastewater from a network of sewer pipes, which in turn collects the wastewater from individual homes and businesses throughout the municipality—as shown in Figure 1. In addition to managing nutrients at the centralized treatment plant (city-scale), nutrients can also be managed at (1) individual buildings, households, and farms (building-scale), as well as (2) neighborhoods or small collections of buildings, households, and farms (community-scale)—as shown in Figure 2. Building-scale and community-scale technologies could be implemented to retrofit existing systems, or could be implemented in new housing developments to maintain current hydraulics and water quality at the centralized plant.

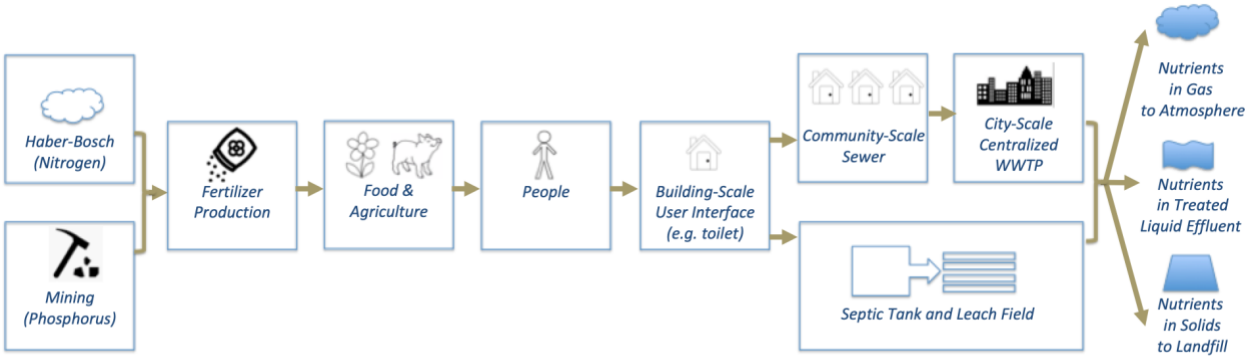

Figure 1. Historically, nutrients have been managed in a linear fashion. Urban wastewater treatment plants meet regulations for liquid effluent of wastewater treatment plants by removing nutrients, whereas septic tanks and leach fields often do not need to meet any nutrient regulations prior to discharge. All the while, nitrogen is generally being moved from the gas phase to the liquid phase and phosphorus is generally being moved from the solid phase to the liquid phase.

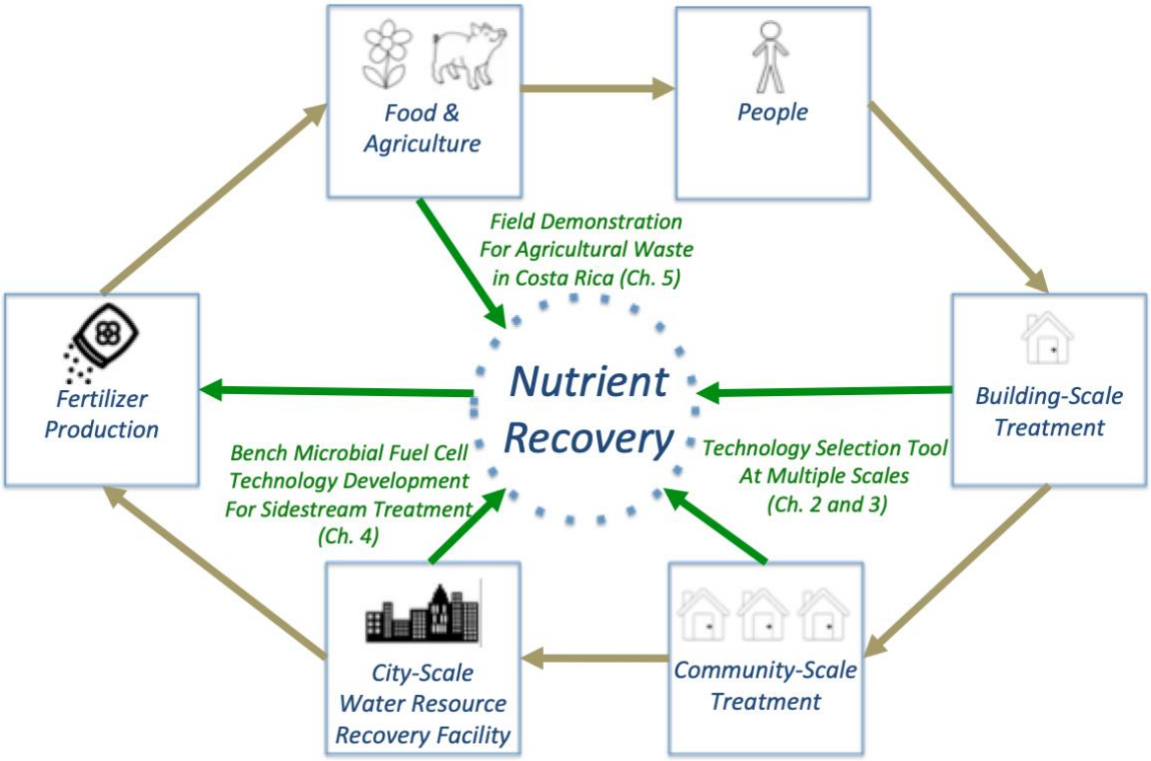

Figure 2. Nutrients can be managed locally and globally for a variety of influents (agricultural wastewater, domestic wastewater) at a variety of scales (building, community, city) to recover nutrients rather than remove them.

However, it is not yet clear if managing nutrients at one scale is better than the others. Because of the large number of technologies available at multiple scales and a number of competing priorities, it may be difficult for communities to make decisions on which technologies to implement. These decisions often do not consider the recovery of nutrients at the building- or community-scales, where nutrients are more concentrated. For example, a community may need to decide what technology to implement, determine if it's economically feasible, and analyze how an upstream technology may affect downstream management.

In addition to removing or recovering nutrients from human wastewater in urban settings, nutrients can be managed in rural or agricultural contexts utilizing decentralized treatment technologies. For example, in Costa Rica, 79,000 out of 93,000 farms have no treatment for their

agricultural waste (Estadisticas Clave sobre el Estado del Ambiente, Costa Rica, 2015), causing increased levels of nutrients in water bodies receiving the waste (Shahady and Boniface, 2018). Opportunities exist to reduce nutrient contamination and recover nitrogen and phosphorus from agricultural waste in Costa Rica and in other countries that currently lack widespread treatment of such wastes.

1.2 Research Objectives

Therefore, the overall goal of this dissertation is to contribute to the achievement of multiple sustainable development goals through the removal and recovery of nitrogen and phosphorus from a variety of influents at a variety of scales. This will be accomplished through the following specific objectives:

(1) Review nutrient management technologies that are currently available at the building-, community-, and city-scales;

(2) Develop a planning matrix that evaluates the appropriateness of nutrient management technologies at these three scales based on practical characteristics of import to stakeholders and decision-makers;

(3) Evaluate how the introduction of new upstream nutrient management technologies affects the treatment efficiency and economics of nutrient management across a sewershed, using Hillsborough County, Florida, as a case study;

(4) Evaluate the performance of a proof-of-concept process that includes a fixed-film nitrification reactor and two-chambered MFC for energy generation and nitrogen removal from sidestreams at centralized wastewater treatment plants; and

(5) Determine the ability of a treatment system for agricultural waste, comprising a tubular anaerobic digester that is integrated with a low-cost, locally produced struvite precipitation

reactor, to contribute to the achievement of multiple sustainable development goals by removing and recovering nutrients during treatment.

1.3 Dissertation Synopsis

This dissertation is organized into six chapters. This first chapter provides a brief explanation of the project background, research gaps, and research objectives. A House of Quality planning matrix for evaluating wastewater nutrient management technologies at three different scales in a sewershed is presented in Chapter 2, and Chapter 3 includes a case-study application of the House of Quality planning matrix set in Hillsborough County, Florida. Chapter 4 and Chapter 5 are more technology based; Chapter 4 investigates energy recovery and nitrogen management from struvite precipitation effluent via microbial fuel cells, and Chapter 5 investigates energy and nutrient recovery from agricultural waste using small-scale tubular anaerobic digester and a locally sourced struvite precipitation reactor. Chapter 6 presents overall conclusions and recommendations for future work.

CHAPTER 2: A HOUSE OF QUALITY PLANNING MATRIX FOR EVALUATING WASTEWATER NUTRIENT MANAGEMENT TECHNOLOGIES AT THREE SCALES WITHIN A SEWERSHED[1]

2.1 Introduction

A typical municipal wastewater treatment system consists of a centralized treatment plant that receives raw wastewater from a network of sewer pipes, which in turn collects the wastewater from individual homes and businesses throughout the municipality. The geographic area and the populace served by such a network are known as the "sewershed" of the sanitation system (Heidler et al., 2006; Teerlink et al., 2012). Just as a watershed is a geographic area from which all surface runoff drains through a single point of exit, a sewershed is "drained" as the final effluent from the centralized treatment plant. The sewershed and its infrastructure can be divided into three distinct scales: the smallest scale is the individual homes or businesses at which wastewater is generated (building scale); the intermediate scale is the conveyance (sewer) system of a particular neighborhood or sub-region of the overall sewershed (community scale); the largest scale (city scale) is the centralized treatment plant and the sewershed that ultimately drains through that centralized plant.

Historically, the main purposes of such sanitation systems have been twofold: to protect human health by minimizing human contact with fecal pathogens, and to protect environmental

[1] Reprinted with permission of Mary Ann Liebert, Inc. Publishers. Kevin D. Orner, Onur Y. Ozcan, Daniella Saetta, Treavor H. Boyer, Daniel H. Yeh, Damann Anderson, and Jeffrey A. Cunningham. *Environmental Engineering* Science, http://doi.org/10.1089/ees.2017.0016. Published in Volume 34, Issue 11. November 1, 2017. Online Ahead of Print: May 25, 2017. Permission is included in Appendix A.

and ecological health by removing chemicals of concern prior to discharging wastewater into a receiving surface water. Constituents of concern include dissolved organic carbon (which exerts a biochemical oxygen demand in receiving waters) and the nutrients nitrogen and phosphorus (which can cause algal blooms and eutrophication in receiving waters). In most cases, any treatment of the wastewater has been performed solely at the centralized treatment plant, rather than at the point of generation or in the conveyance system, and has consisted solely of *removal* of regulated constituents rather than *recovery* of those constituents in a useful form.

Recently, however, an important paradigm shift has been in progress, and municipal "wastewater" is increasingly viewed not merely as a waste product that requires treatment, but as a resource from which valuable products can be derived (e.g., Guest et al., 2009; Mo and Zhang, 2013; WEF, 2014; Englehardt et al., 2016). Associated with that paradigm shift is a renewed interest in the possibility of on-site or decentralized treatment systems, i.e., treatment systems that would be deployed at either the building scale or the community scale rather than at the city scale (Crites and Tchobanoglous, 1998; Tchobanoglous et al., 2004; Verstraete et al., 2009; Gikas and Tchobanoglous, 2009; The Johnson Foundation at Wingspread, 2014; Roefs et al., 2017). Technologies across all three scales can be combined to form a distributed system. Therefore, the term sewershed encompasses buildings which are currently on sewer but may move to a decentralized treatment system in the future but also buildings which could reasonably be connected to a sewer but currently use decentralized technologies that may not necessarily include a sewer (e.g. a suburban home using a septic system) (WERF, 2011). Therefore, two important and interrelated questions have recently arisen: First, can nitrogen and/or phosphorus be economically recovered from wastewater in a useful form, rather than merely removed from

the wastewater and discarded? Second, how should we best manage nitrogen and phosphorus at all three scales, not just at the city scale?

Many technologies already exist to remove and/or recover nitrogen and phosphorus from wastewater, and many others have been proposed or are in different stages of technical development and maturity (Morse et al., 1998; Yeoman et al., 1998; Doyle and Parsons, 2002; de-Bashan and Bashan, 2004; Cai et al., 2013; WEF, 2014). However, it must be recognized that the applicability or appropriateness of a candidate nutrient removal/recovery technology depends on the scale at which it is to be deployed. A technology that is appropriate for nutrient removal/recovery at a large centralized treatment plant, for instance, may not be appropriate at an individual household. This scale dependence of candidate technologies arises from a number of considerations, such as availability of space, aesthetics, training or expertise of the system operators (e.g., a licensed wastewater treatment plant operator versus an individual homeowner), economies of scale, etc. Furthermore, different municipalities or utilities (or even different sewersheds within a single municipality) may have different priorities, also due to a number of possible factors – different regulatory constraints, different concerns of the citizens or stakeholders in the municipality, etc. Therefore, what is needed is a systematic framework or methodology for assessing candidate nutrient management technologies that accounts for scale of application. That framework should also be flexible enough to account for the differing needs or priorities of different municipalities or utilities. There have been a few previous efforts towards this objective via life cycle assessment of nutrient management technologies (e.g., Lundin et al., 2000; Cornejo et al., 2016); however, life-cycle analysis prioritizes environmental sustainability without accounting for certain practical factors (cost, aesthetics, technical maturity, etc.) that are essential to most stakeholders and decision-makers.

Therefore, the objectives of this chapter are (1) to review nutrient management technologies that are currently available at the building, community, and city scales and (2) to develop a planning matrix that evaluates the appropriateness of nutrient management technologies at these three scales based on practical characteristics of import to stakeholders and decision-makers. Because each sewershed is unique, the planning matrix developed herein should be customizable to account for the priorities of any given municipality, thereby representing a flexible and important tool for sewershed-scale nutrient management. In the current chapter, I target nitrogen and phosphorus specifically, but my aim is to develop a framework that is flexible enough that it could, in the future, be expanded to also consider organic nutrients or other recoverable resources.

Towards these goals, the planning matrix developed herein is based on the House of Quality (HoQ) structure, which is a Quality Function Deployment method typically used in commercial businesses to determine how well a product meets the needs of its customers (Hauser and Clausing, 1988; Park and Kim, 1998). The planning matrix developed to evaluate the appropriateness of nutrient management technologies is based on the HoQ structure (Hauser and Clausing, 1988). Quality Function Deployment started in Japan in the 1970s at Mitsubishi and Toyota (Hauser and Clausing, 1988) with the goal of implementing a tool that determines how well a company's product meets the needs of its customers. The HoQ came to the United States in the later 1980s to companies such as Ford, Xerox, General Motors, Campbell's Soup, Colgate, and Fidelity Trust and is used today across several disciplines (Griffin and Hauser, 1993; Ho et al., 2012). However, to the best of our knowledge, the HoQ has not been previously used to evaluate wastewater treatment technologies.

2.2 Methods

2.2.1 Planning Matrix Structure

The structure of the HoQ, as it is applied here, is shown in Figure 3. The house has five regions, which are elaborated in the following subsections.

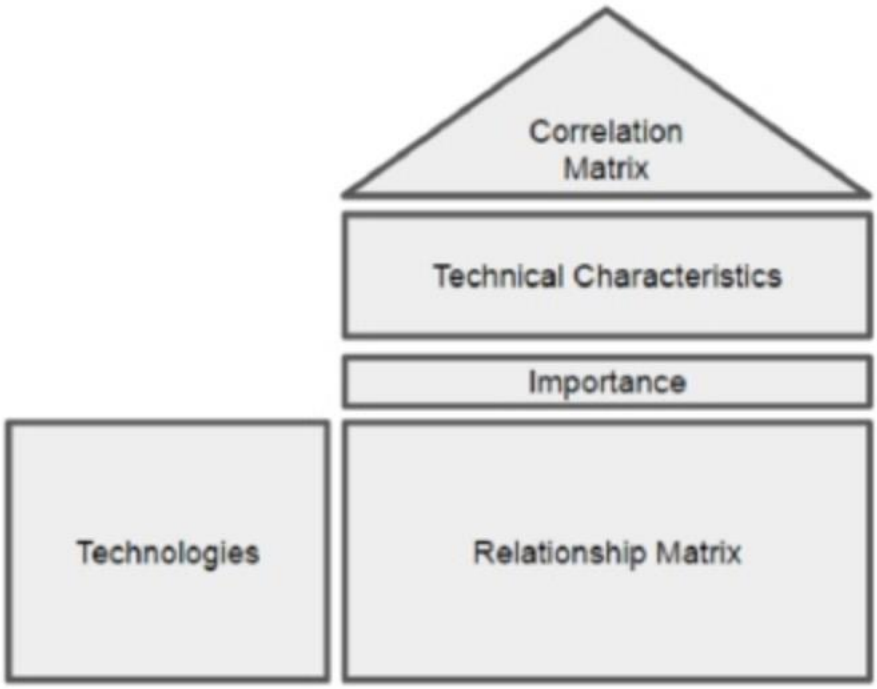

Figure 3. Modified version of House of Quality used to evaluate nutrient removal and recovery technologies at the building, community, and city scales (adapted from Lowe, 2000).

The left wall, "Technologies", is a list of all potential technologies available for nutrient removal or recovery at a given scale. The ceiling, labeled as "Technical Characteristics", presents the attributes of candidate technologies believed to be important to the client. In the current analysis, we have selected ten characteristics, but the developer of the HoQ may choose as many characteristics as are relevant to the stakeholders. "Importance" is the weighting given to each technical characteristic. The central part of the HoQ is the "Relationship Matrix", where the technologies are evaluated for each of the technical characteristics. The roof of the house, the

“Correlation Matrix”, displays the interaction between the technical characteristics (i.e., some of the technical characteristics are likely to be correlated – a technology that scores poorly in “materials consumption” is likely to also score poorly in “cost”, but other technical characteristics are uncorrelated – how a technology scores in “aesthetics” is not likely to provide any information about how it will score in “technical maturity”).

2.2.2 Technologies

A list of nutrient management technologies available at each of the three scales was compiled through a review of literature. As described subsequently, the HoQ enables each technology to be evaluated according to the selected technical characteristics and the selected weighting for each characteristic.

2.2.3 Technical Characteristics

The technologies on the left wall were evaluated by ten relevant technical characteristics chosen to cover the main attributes that are important for nutrient removal and recovery from wastewater. Operational definitions for each technical characteristic are elaborated in Table 1. The ten technical characteristics are based on expert opinion and supported by the literature. In addition, HoQ methodology is flexible, and a different user could select different characteristics if those employed here do not account for some other important aspect or consideration.

Table 1. Abbreviations and general description for each of the ten technical characteristics.

	Technical Characteristic	Abbre-viation	General Description
1	Ease of operation and maintenance	O&M	The lack of complexity and resources needed in the technical handling of the technology after it has been installed
2	Size/footprint	SF	Extent to which the technology requires physical space
3	Capital cost	CC	Cost to install and ready the technology for use, as compared to the 'baseline' technology of sewer connections with centralized treatment
4	Operational cost	OC	Extent to which the product requires operational input in terms of energy, chemicals, water, and labor
5	Value of end products	EP	Extent to which the technology provides valuable products, such as nutrients, energy, water, and commodities
6	Technical maturity	TM	Readiness of a technology to be effectively used in nutrient removal or recovery as reported in literature
7	Environmental impact[2]	EI	Extent of the impact to the environment on the basis of air pollution, land pollution, release of nitrogen, and release of phosphorus
8	Performance in nitrogen removal	PN	Extent of nitrogen removal or recovery by the technology
9	Performance in phosphorus removal	PP	Extent of phosphorus removal or recovery by the technology
10	Aesthetics	A	Extent to which the technology is acceptable to stakeholders on the basis of visual appeal, noise level, and odors

2.2.4 Importance

Weightings for each characteristic, shown in Table 2, were allocated for each of the three scales based on the judgment of the authors. The different weightings for the different scales reflect the importance of the technical characteristics at a given scale. For instance, ease of

[2]*Pollutants* should be considered as *byproducts, adverse effects, or other harmful impacts*. Also, some technologies may reduce one type of pollutant but not another (e.g. nitrogen but not phosphorus), and may require a second technology to further reduce pollutants prior to discharge.

operation and maintenance is weighted strongly at the building scale because individual homeowners and building managers will only accept and employ a technology that is easy to use and maintain; operation and maintenance is weighted less strongly at the city scale because a centralized treatment plant has trained professional operators who are dedicated to managing the adopted technologies. At each of the three scales, the average weighting for the ten characteristics is 1.0.

Table 2. Weightings for each scale for the ten technical characteristics.

		Weighting		
		Building Scale	**Community Scale**	**City Scale**
1	O&M	1.5	1.3	0.8
2	SF	1.4	1	0.6
3	CC	1	1	1.1
4	OC	1	1.1	1.1
5	EP	0.5	0.8	1.1
6	TM	1.2	1.1	1.1
7	EI	0.7	1	1.3
8	PN	0.7	0.9	1.2
9	PP	0.7	0.9	1.2
10	A	1.3	0.9	0.5

The importance of one technical characteristic over another for a given assessment can vary greatly depending on the economic and geographical context of the municipality or community needs and priorities; thus, the importance can be modified by the WWTP stakeholders using different weighting. For example, if cost is an overriding consideration for a

particular municipality, that municipality could provide a greater weighting for operational cost and concomitantly reduce the weighting for the other characteristics.

2.2.5 Relationship Matrix

The relationship matrix reports how well each technology fulfills the technical characteristics chosen by the stakeholder (in this case the authors). In this chapter, literature was reviewed for each technology and its associated ten technical characteristics. The technologies were evaluated according to the rubric for the ten technical characteristics in Table 3. Technologies that performed well on a given technical characteristic (according to the judgment of the authors) received a "+" or "++" for that characteristic while technologies that did not perform well received a "–" or "––". For example, a review of literature indicates that sidestream chemical precipitation[3] may recover 90% of phosphorus, so according to the rubric in Table 3 it would receive a "++" for performance in phosphorus removal in Table 7. All of the technical characteristics except for the capital cost use an absolute (rather than relative) metric to independently determine the technology's score for that characteristic. The capital cost is scored relative to a *baseline scenario* of centralized wastewater treatment, which includes flush toilets, combined sanitary and storm sewer systems, gravity sewer, and conventional wastewater treatment. Conventional wastewater treatment is defined as activated sludge, a secondary wastewater treatment with the top priority of carbon removal (and minor nutrient removal in the sludge), but no tertiary treatment of nitrification and denitrification.

[3] In Chapter 2 chemical precipitation is specifically meant as struvite precipitation.

Table 3. Rubric used to evaluate each nutrient removal and recovery technology using ten technical characteristics.

Technical Characteristic	Metric	++	+	0	-	--
Ease of Operation & Maintenance	Amount of oversight needed and complexity and input of resources	minimal oversight needed; very low complexity and input of resources	little oversight needed; low complexity and input of resources	some oversight needed; moderate complexity and input of resources	significant oversight needed; high complexity and input of resources	very significant oversight needed; very high complexity and input of resources
Size/ Footprint	Hydraulic retention time of the technology	seconds	minutes	hours	days	weeks
Capital Cost	Estimated fixed, one-time cost to install and ready the technology for use (as compared to the baseline scenario)	Estimated to be better than 20% less expensive (for same design capacity) than baseline technology	Estimated to be 10-20% less expensive than baseline technology	Estimated to be within 10% of the baseline cost	Estimated to be 10-20% more expensive than baseline technology	Estimated to be worse than 20% more expensive than baseline technology
Operation and Maintenance Costs	Number of inputs (energy, chemicals, water, and/or labor) needed to operate and maintain the technology[4]	0 inputs	1	2	3	4+
Value of End Products	Number and quality of products (nutrients, energy, water, and commodities) recovered by a technology	Creates multiple products or one product of very high quality	Creates two products or one product of moderate quality	Creates one product of marginal products	Creates no valued products	Creates burden product
Technical Maturity	Most advanced status reported in literature	Widespread full-scale operation	Limited full-scale operation	Pilot location	Laboratory experiments	No laboratory experiments
Environmental Impact	Number of possible adverse effects to the environment caused by the implementation of the technology	0 pollutants[5]	1 pollutant	2 pollutants	3 pollutants	4 pollutants
Performance of N Removal	Percentage removal or recovery of nitrogen because of the technology	90+	75-90	50-75	25-50	0-25
Performance of P Removal	Percentage removal or recovery of phosphorus of the technology	90+	75-90	50-75	25-50	0-25
Aesthetics	Number of senses offended (sight, sound, smell) by a technology	Enhances aesthetics	0	1	2	3

[4] Inputs can be considered as expenditures of time and money. Multiple types of labor should be considered separately—administrative labor, laboratory labor, field labor, etc.

[5] *Pollutants* should be considered as *byproducts, adverse effects, or other harmful impacts*

Each technology can then be awarded an overall technology score by awarding five points for "++", 4 points for "+", etc. The score in each category is multiplied by the importance or weighting factor assigned to that category (described above). The overall technology score is the sum of the weighted scores for the ten technical characteristics as shown by the equation below:

$$Overall\ score = Score_1 Weighting_1 + Score_2 Weighting_2 + [\ldots] + Score_{10} Weighting_{10}$$

A higher overall numerical score indicates a higher-performing technology. In Section 3, the overall numerical score for each technology is compared to the overall numerical score of the baseline scenario mentioned above, allowing us to identify nutrient management alternatives at each scale that might be competitive with, or preferable to, the current technology standard in the context of the priorities of the authors.

Overall numerical scores depend on both the technology's score in each characteristic and the weighting assigned to each characteristic. Both of these can be context-specific. Weightings are context-specific because different stakeholders or municipalities may have different priorities; in some cities, for example, cost may be an overriding concern, but in others, cost may be secondary to environmental impact or other factors. Furthermore, a technology's score within each characteristic is also context-specific; in regions where population density is low, for instance, decentralized or on-site systems (e.g., septic systems) should score better than in regions where population density is high. Because of this dependence on context, the current chapter attempts to base its analysis on a "generic" city or municipality, but it must be recognized that the analysis herein is predicated on a set of built-in assumptions, such as the housing density being high enough for conventional centralized treatment to be economically viable.

For each scale, the “best” two to four technologies with the highest overall numerical scores are discussed in depth in section 2.3. To select the technologies that are “best” at each scale, we looked for total scores greater than 30, and for a gap of at least 8% between the scores of the “best” technologies and the other technologies considered.

2.2.6 Correlation Matrix

The roof of the HoQ displays reinforcing interactions between technical characteristics with a checkmark and balancing interactions with an “x”. For example, size/footprint and capital cost would have a checkmark because a larger sized facility (higher size/footprint) would require more money to purchase the property (higher capital cost). In contrast, a more mature technology is more likely to have a reduced capital cost because its history of use could allow for many organizations to design and construct such a technology; this would be indicated by an x at the intersection of operation and maintenance and capital cost.

2.3 Analysis of Treatment Technologies

2.3.1 Building-Scale House of Quality

In developed countries, homes and buildings are equipped with flush toilets that typically use potable water to convey the nutrient-rich waste stream to treatment facilities far outside of the land boundaries within which the building stands. Having treatment processes within the buildings’ boundaries provides the opportunity to intercept wastewater where it has the highest nutrient concentrations and lowest volume. More specifically, diverting urine at the point of collection has been studied as a possible solution to reduce nutrient loading to the centralized WWTP (Jimenez et al., 2015). Urine is estimated to contribute 75% of the nitrogen mass load and 50% of the phosphorus mass load to a WWTP, while only contributing 1% of the flow by volume (Larsen and Gujer, 1996).

Table 4 is the House of Quality that compares the baseline technology of a conventional toilet (connected to a sewer system and, eventually, a centralized treatment plant) to several other building-scale nutrient management technologies, including struvite precipitation from urine, aerobic and anaerobic membrane bio-reactors (MBR), treatment wetlands, and on-site wastewater treatment systems (OWTS), commonly called septic systems (Crites and Tchobanoglous, 1998). Table 4 includes technologies that treat combined wastewater (graywater and blackwater), blackwater, or diverted urine. Each of these waste streams has a unique composition and requires specific treatment mechanisms that are important considerations (Rashidi et al., 2015).

Using the House of Quality, a numerical score was calculated for each building-scale technology, as described in Section 2.5. Based on the weightings employed here, conventional wastewater treatment scored the highest (33.9). Two other technologies had scores above 30: composting toilets and aerobic MBRs. Composting toilets, which received a score of 30.8, aerobically treat human waste to create nutrient-rich compost that can be used as a soil amendment in agricultural operations (Anand and Apul, 2014). Composting toilets achieved a relatively high overall score due to theoretically high nutrient recovery, production of a useable product with no pollutants to air, land or water, and only a slight increase in operational cost. As a result of these characteristics, composting dry toilets have been used in remote locations, such as parks, as a method of managing wastewater. Aerobic MBRs received the third highest score, 30.0, in Table 4. At the building scale, aerobic MBRs can treat wastewater to reclaimed-water quality. These systems have been placed inside “green” buildings for the ability to produce reclaimed water by removing nutrients on a relatively small footprint (Rashidi et al., 2015). This

technology received a “++” in the following categories: end products, performance in N removal, and performance in P removal.

The top score of conventional wastewater treatment highlights the difficulty of bringing nutrient recovery technologies into the building scale when conventional wastewater treatment is the status quo. Conventional treatment is easy for the user to maintain (i.e., just a toilet and home plumbing connected to a sewer main), has a small footprint, and is not offensive aesthetically. The technical characteristics with the highest weighting at the building scale are operation and maintenance (1.5), size/footprint (1.4), and aesthetics (1.3); only conventional wastewater treatment scored positively across all three of these technical characteristics. The technologies that scored most closely to the baseline technology are those that have high performance in removal or recovery of nitrogen, have high performance in removal or recovery of phosphorus, and have low impacts to the environment. The composting toilet had the second highest score despite having negative scores in operation and maintenance, size/footprint, and aesthetics. Technologies that did not perform well in this evaluation not only had negative scores in operation and maintenance, size/footprint, and aesthetics, but also in technical maturity and several other technical characteristics.

Table 4. House of Quality for building-scale nutrient management technologies.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Importance		1.5	1.4	1.0	1.0	0.5	1.2	0.7	0.7	0.7	1.3	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/ Density/ Footprint	SF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	Conventional Wastewater Treatment		++	+	0	+	--	++	--	--	--	+	33.9
	Composting Toilet**		-	--	0	+	0	+	++	++	++	-	30.8
	Aerobic MBR		--	0	--	0	++	+	0	++	++	0	30.0
	Treatment wetlands		-	-	--	+	-	+	+	-	-	++	28.7
	Septic systems		+	-	-	0	--	++	--	--	--	+	27.6
	Direct Urine Application as Fertilizer**		--	--	--	+	+	0	++	++	++	-	26.6
	Anaerobic MBR		-	0	--	0	+	0	0	--	--	0	24.2
	Nitrification and Distillation of Urine**		--	--	--	0	0	-	+	++	++	-	23.2
	Struvite + Absorption with Zeolites in Urine**		--	--	--	-	+	-	0	+	+	--	19.3
	Urine ANAMMOX**		--	--	--	0	0	-	+	++	--	--	19.1
	Struvite Precipitation from Urine**		--	--	--	-	++	-	+	--	++	--	19.1
	NH3 Stripping to H2SO4 in Urine**		--	--	--	-	+	-	+	++	--	--	18.6
	Anaerobic Digestion		-	--	--	0	0	+	-			--	17.4
	Anion Exchange in Urine**		--	--	--	--	0	-	0	--	++	--	16.4

Note: an "**" indicates source separation of urine

2.3.2 Community-Scale House of Quality

Community-scale technologies have the benefit of treating wastewater in moderate volume while often being closer to the waste source (and/or potential reuse location) than a centralized treatment plant. Community-scale technologies predominantly incorporate technologies that require minimal maintenance and oversight, incur low operational costs, and

are technically mature. Most of the technologies listed in the community-scale House of Quality shown in Table 5 use physical and biological treatment processes to remove contaminants (Makropoulos and Butler, 2010; Massoud et al., 2009; *Onsite Wastewater Treatment Systems Manual*, 2002). An activated sludge system was used as the baseline technology to compare the capital costs.

Using the House of Quality, a numerical score was calculated for each community-scale technology, as described in Section 2.5. Based on the weightings employed here, constructed wetlands (38.0) scored the highest, followed by facultative lagoons (32.4), rotating biological contactors (32.3) and biological nutrient removal (31.9). Constructed wetlands are artificially engineered wetlands treating wastewater via processes involving uptake by vegetation, soil absorption, sedimentation, and microbial activity. The preference towards constructed wetlands is primarily due to its high scores in operation and maintenance, operational costs, technical maturity, and aesthetics. Facultative lagoons utilize layers with different dissolved oxygen levels to treat wastewater without mechanical mixing or aeration. Facultative lagoons are not as aesthetically pleasing as constructed wetlands. Biological nutrient removal (BNR), defined here as the removal of N and P using a combination of nitrification, denitrification, and enhanced biological phosphorus removal processes, may be implemented on its own or in combination with different types of reactor systems such as membrane bioreactors (MBRs). BNR technologies create moderate- to high-quality reclaimed water. Biological nutrient removal and MBRs have high scores in end products, technical maturity, nitrogen performance, and phosphorus performance.

Septic systems, which are deployed widely in cases where conventional treatment is not viable (e.g., low population density), received a lower score (27.6) than other building-scale

technologies such as compositing toilets (30.8), aerobic MBRs (30.0), and treatment wetlands (28.7). Septic systems have lower scores than the three alternative technologies for the technical characteristics of end products, environmental impact, nitrogen performance, and phosphorus performance. Therefore, in cases where a high degree of decentralization is required, these three technologies may be preferable.

Constructed wetlands, which achieved the top score by several points, and facultative lagoons separated themselves from the competition based on cost (high scores in operation and maintenance and operational costs). Accordingly, these two technical characteristics were deemed to have the most importance at the community scale (along with technical maturity, which showed significantly less variation in scores). However, constructed wetlands, facultative lagoons, nor rotating biological contactors easily facilitate recovery of N or P in a readily usable form. Biological Nutrient Removal was competitive because of its high scores in end products, nitrogen performance, and phosphorus performance. If the future brings increased demand for nutrient recovery, along with technical advances in making community-scale technologies easy and cost effective to operate and maintain, it may be preferable to install rotating biological contactors, BNR or MBR technologies rather than the “low-tech” but cost-effective wetlands or lagoons.

Table 5. House of Quality for community-scale nutrient management technologies.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Importance		1.3	1.0	1.0	1.1	0.8	1.1	1.0	0.9	0.9	0.9	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SF	✓										
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM			✓								
	Environmental Impact	EI		✓									
	Performance N	PN							✓				
	Performance P	PP											
	Aesthetics	A		✓									
Technologies	Constructed Wetlands		++	--	++	++	-	++	0	0	0	++	38.0
	Facultative Lagoons		++	--	+	+	--	++	0	+	-	-	32.4
	Rotating Biological Contactor		-	0	-	-	+	++	0	++	+	0	32.3
	Biological Nutrient Removal (BNR)		--	0	-	-	+	++	0	++	++	0	31.9
	Algal Photobioreactors		-	-	-	-	+	0	+	++	++	0	31.0
	Sequencing Batch Reactors - BNR mode		--	0	-	-	+	+	0	++	++	0	30.8
	Communal Septic Systems		++	-	+	++	--	++	-	--	--	0	30.8
	Recirculating Sand Filter		0	-	0	+	--	++	0	0	-	+	30.7
	Intermittent Sand Filter		+	-	0	+	--	++	0	-	-	0	30.2
	Membrane Bioreactors - BNR mode		--	0	--	--	++	+	0	++	++	0	29.5
	Upflow Anaerobic Sludge Blanket		-	0	0	++	+	+	0	--	--	0	29.2
	Aerated Lagoons		+	--	0	+	--	++	0	0	--	0	29.2
	Activated Sludge Systems		-	0	0	0	0	++	0	-	-	0	29.1
	Anaerobic Lagoons		++	--	+	++	--	++	-	--	--	-	28.9
	Algal Membrane Bioreactors		-	-	--	--	++	-	+	++	++	0	28.6
	Anaerobic Membrane Bioreactors		-	0	--	+	+	0	0	--	--	0	25

2.3.3 City Scale: Mainstream House of Quality

City-scale technologies for mainstream wastewater treatment are common in urban settings due to their ability to treat large volumes of water in one central location. To protect the health of ecosystems that receive treatment plant discharge, mainstream technologies reduce the environmental impact of collected sewage by removing carbon, nitrogen, and phosphorus. Several mainstream technologies are evaluated in Table 6 using the House of Quality. To evaluate capital costs, technologies are compared against a baseline scenario of activated sludge for carbon removal, separate-stage nitrification/denitrification for nitrogen removal, and alum addition for phosphorus removal; technologies in Table 6 that remove only one element (C, N or P) are compared against the relevant treatment process that removes that element.

A large number of mainstream technologies are available, but many of these technologies are similar in principal and operation, varying mainly in configurational details; therefore, in Table 6, similar technologies are grouped together and not all candidate technologies are included. For example, anaerobic/anoxic/oxic (A^2O) treatment is included, but anoxic/oxic (A/O) and Modified Ludzack-Ettinger (MLE) processes are not included, because A^2O can be considered a combination of A/O and MLE. Similarly, fixed-film nitrification-denitrification includes both trickling filters and rotating biological contactors (RBCs).

Table 6. House of Quality for city-scale mainstream nutrient management technologies.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Importance		0.8	0.6	1.1	1.1	1.1	1.1	1.3	1.2	1.2	0.5	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	A^2O		++	0	-	0	-	++	+	++	++	0	37.7
	Oxidation Ditch		-	0	+	0	-	++	+	++	++	0	37.5
	5-stage Bardenpho		+	0	-	0	-	++	+	++	++	0	36.9
	Membrane Technologies		-	0	--	-	-	++	+	++	++	0	33.1
	Chemically Enhanced Primary Treatment		+	+	++	-	-	+	0	+	-	0	32.5
	Fixed Film Nitrification/ Denitrification		-	0	+	0	-	++	0	++	--	0	31.4
	Conventional Separate-Stage Nitrification/ Denitrification		0	0	0	-	-	++	+	++	--	0	31.3
	Shortcut Nitrogen Removal		+	0	0	-	-	++	+	++	--	0	30.6
	Chemical Precipitation and Crystallization		0	0	0	-	-	++	0	--	++	0	30
	Mechanical Separation		+	+	++	0	-	++	-	--	--	0	28.6
	Enhanced Biological Phosphorus Removal		-	0	0	-	-	++	0	--	+	0	28
	ANAMMOX		-	0	-	-	-	0	+	+	--	0	26
	Sludge Ash Recovery		-	0	-	-	++	-	0	--	0	0	25.7

Based on the weightings employed here, the three highest-scoring technologies were A^2O (37.7 points), Oxidation Ditch (37.5 points), and 5-stage Bardenpho (36.9 points). The A^2O process achieves high removal of both nitrogen and phosphorus by placing an anaerobic chamber before the anoxic and aerobic chambers. It receives high scores for O&M, TM, EI, PN, and PP.

The Oxidation Ditch receives influent in its anaerobic reactor, which is followed by a rotating ditch that alternates between anoxic zones and mechanically-mixed aerobic zones. The oxidation ditch scored highly in CC, TM, EI, PN, and PP. The 5-stage Bardenpho utilizes additional anoxic and aerobic reactors to meet requirements of low TN and low TP and has similar scores to A^2O.

In contrast to the baseline technology of separate stage nitrification-denitrification, all three of the highest-scoring technologies make use of carbon that is already present in the wastewater to drive denitrification, thereby saving money on operation and maintenance by not needing an external carbon source. Candidate technologies for city-scale mainstream treatment that did not perform well in this evaluation are those that do not recover valuable resources, lack maturity, and/or require significant amounts of money, chemicals, or energy to operate and maintain.

2.3.4 City Scale: Sidestream House of Quality

During city-scale mainstream treatment, anaerobic digestion is often employed to treat the sludge from primary and secondary treatment. Effluent of anaerobic digestion includes biogas, biosolids, and a liquid effluent stream that is typically recycled back to the beginning of the mainstream treatment process. This liquid effluent stream is often called the sidestream. Compared to the mainstream, the sidestream contains higher concentrations of nitrogen and phosphorus, lower flow rates, and lower levels of carbon. Therefore, mainstream technologies that rely on higher COD:N ratios for denitrification, such as A^2O, Bardenpho, and oxidation ditch, are not considered for sidestream treatment. A baseline scenario for sidestream treatment is return to the headworks without additional treatment; treatment of the sidestream is presently increasing in popularity but is not yet commonly used.

Several sidestream technologies are evaluated in Table 7 using the House of Quality. The two top-scoring technologies were ion exchange and chemical precipitation and crystallization. Ion exchange, which had a score of 36.1, recovers nutrients that can be used as fertilizer in the sidestream, a more favorable location for recovery (as compared to mainstream treatment) due to higher concentrations of nitrogen and phosphorus. One example of an ion exchange technology is the RIM-NUT process, which uses the natural zeolite clinoptilolite for ion exchange and a strong base resin for regeneration and subsequent reuse (Liberti et al., 1986). The process recovers 90% of nitrogen, which can be used for fertilizer. Ion exchange can be integrated with chemical precipitation to recover both nitrogen and phosphorus. Negative scores are given for operation and maintenance, capital cost, and operational cost, mostly due to the cost of resin and materials. "++" scores were given for end products, environmental impact, nitrogen performance, and phosphorus performance. Recovering N and P in the sidestream offers the dual benefits of producing a potentially valuable product and reducing the cost of removing N and P during mainstream treatment (which would otherwise be required if a nutrient-rich sidestream is returned to the plant headworks).

Table 7. House of Quality for city-scale sidestream nutrient management technologies.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Importance		0.8	0.6	1.1	1.1	1.1	1.1	1.3	1.2	1.2	0.5	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	Ion Exchange		-	+	--	-	++	0	++	++	++	0	36.1
	Chemical Precipitation and Crystallization		0	+	-	-	++	++	+	-	++	0	35.3
	Ammonia Stripping		0	0	-	-	++	+	+	++	--	0	32.4
	No Treatment		-	++	0	++	-	++	++	--	--	+	32.0
	Forward Osmosis		0	0	--	-	-	-	++	++	++	0	31.9
	Conventional Nitrification-Denitrification		0	0	-	-	-	++	+	++	--	0	30.2
	Microbial Fuel Cell		0	0	--	0	0	0	+	++	--	0	29.1
	Short-cut Nitrogen Removal/Recovery (ANAMMOX, SHARON, CANON, SHARON-ANAMMOX, DEAMOX)		-	0	-	-	-	++	+	+	--	0	28.2
	Breakpoint Chlorination		-	-	-	-	--	+	+	+	--	0	25.4

Chemical precipitation and crystallization, which uses the addition of a divalent or trivalent metal salt to remove phosphorus (and, to a lesser extent, nitrogen) through sedimentation of the precipitate (Halling-Sorensen, 1993; Jenkins et al., 1971), received a score of 35.3. Cations such as calcium, iron, and aluminum can be added to bind with phosphate within a fluidized reactor to be settled and recovered. However, the addition of Fe^{3+} or Al^{3+} may not

produce a recoverable product that has a market to be sold. If Mg^{2+} is added, as in the case of struvite precipitation, a fertilizer product ($MgNH_4PO_4$) can be recovered. Struvite precipitation typically recovers 80–95% of the phosphorus in the sidestream; however, only 10–40% of the nitrogen is typically recovered during the process (WERF, 2012). Using the example of struvite precipitation, this technology receives "++" scores in end products and technical maturity due to recovering the engineered struvite precipitate and the subsequent reduction in likelihood of nuisance struvite precipitation. However, it receives negative scores in capital cost, operational cost, and nitrogen performance primarily because of the chemical addition. Companies that offer a process to produce a struvite product include Aquatec Maxcon (Crystalactor®) and Ostara. Engineered struvite precipitation reduces the potential for nuisance struvite precipitation and, as with ion exchange, also reduces the amount of nutrients needing treatment in the mainstream.

Overall, sidestream technologies such as ion exchange and chemical precipitation and crystallization scored the highest because of their ability to remove nitrogen while also recovering valuable end products that can be used as a fertilizer.

2.4 Discussion

2.4.1 Expected Future Trends

The rankings at each scale generally align with current wastewater treatment practice. For instance, at the building scale, conventional treatment is ranked highest because of its easy maintenance, small footprint, and inoffensive aesthetics. Similarly, at the city scale, top-ranked technologies are those commonly employed (e.g. A^2O, oxidation ditch) that use the dissolved organic carbon present in the wastewater to drive denitrification.

However, future trends will likely affect the technologies, weightings, and scores and therefore change the ranking of the technologies. One trend is diminishing phosphorus reserves,

which are likely to be depleted in the next 50-100 years (Cordell et al., 2009). While the supply is diminishing, phosphorus demand is expected to increase until it reaches its peak demand around 2030 (Cordell et al., 2009). This trend is exacerbated by the uneven global distribution of the phosphorus reserves. Thus, value of end products would receive higher weightings in the future as P recovery becomes more important and provides more revenue. Another trend is continued research and development of wastewater treatment technologies, which may result in higher scores across all ten technical characteristics for up-and-coming technologies. Additionally, the implementation of test beds can provide opportunities to improve the technical characteristics of developing technologies while minimizing risk for municipalities (WERF, 2017). The trend of continued research and development of wastewater treatment technologies is especially noticeable at the building and community scales, where several publications highlight the need for and development of source separation and decentralization technologies (Larsen et al., 2013; Skambraks et al., 2017). As these technologies develop and become easier to operate and maintain, reduce in size, and improve aesthetically, they will challenge the current paradigm of a flush toilet connected to a septic tank or a sewer system. Building- and community-scale technologies, because of their more decentralized nature, are more nimble and can produce cost savings due to reduced idle capacity when population growth is less than predicted (Roefs et al., 2017). A path forward is distributed systems, which combine both centralized and decentralized treatment technologies.

2.4.2 Limitations of This Analysis

One limitation of this chapter is that the weightings cannot be universally applied to every community. Scores were assigned to each technology and weightings applied to each technical characteristic without any specific community in mind, thus the analysis is somewhat

generic. Therefore, each city's particular context might override the scores and weightings assigned based on its unique geography, financial resources, population, and preferences. For example, some centralized plants may be located in areas where the residents are highly concerned about aesthetics and would not choose any technology that creates odors. However, our easily customizable planning matrix can accommodate a situation such as this because each community can adjust the weightings of the ten characteristics (e.g. aesthetics) to determine the most appropriate city, community, and building scale nutrient management technologies.

Another limitation of this chapter is that the ramifications of new upstream treatment on existing downstream treatment is unknown. The introduction of building-scale and community-scale technologies could produce a number of consequences, such as reduced nutrient loading on existing city-scale wastewater treatment plants or reduced costs of centralized treatment. However, the reduced nutrient loading could prevent a centralized plant from economically precipitating and selling struvite. Therefore, future research is needed that evaluates how the introduction of new upstream nutrient management technologies affects the treatment efficiency and economics of nutrient management across an entire sewershed.

2.4.3 Relation to Other Decision-Making Tools

The House of Quality planning matrix described herein is an efficient tool for because Tables 4-7 -- which list all technologies, their grades across ten technical characteristics, and a "default" importance factor -- can be easily adjusted to accommodate different contexts. This method is complementary and can be used in conjunction with, or instead of, other methods such as the eco-balance (e.g., Kimura and Hatano, 2007), life-cycle assessment (e.g., Cornejo et al., 2016), stakeholder analysis (e.g., Lienert et al., 2013) and multi-criteria decision analysis (e.g., Flores et al., 2008). Eco-balance, which identifies the environmental impact of various business

activities, and life-cycle assessment, which identifies and analyzes the environmental impact of various processes or products, provide depth on one technical characteristic such as environmental impact, but do not provide the breadth of ten technical characteristics. Stakeholder analysis identifies, prioritizes, and understands key stakeholders, but doesn't provide decision-making support. The House of Quality utilizes elements of multi-criteria analysis such as defining the decision, identifying stakeholder interests, weighting stakeholder interests, and scoring alternatives. The planning matrix developed here is unique in that it utilizes ten technical characteristics based on expert opinion and supported by literature and evaluates wastewater treatment technologies based on their scale of application.

CHAPTER 3: A CASE STUDY FOR ANALYZING NUTRIENT MANAGEMENT TECHNOLOGIES AT THREE SCALES WITHIN A SEWERSHED

3.1 Introduction

A centralized wastewater treatment plant typically collects wastewater from homes and businesses via a network of sewer pipes. The people and geographic area that the network serves is known as a "sewershed" (Heidler et al., 2006; Teerlink et al., 2012), analogous to a "watershed" that is drained by a network of tributaries feeding into a main river. A sewershed can be considered to comprise three distinct scales: 1) building scale, where individual homes and businesses generate wastewater; 2) community scale, where the particular neighborhood or sub-region of the overall sewershed conveys wastewater; and 3) city scale, which consists of a centralized treatment plant that receives the wastewater from the entire sewershed (Figure 4; see also Chapter 2).

A developing paradigm in wastewater treatment is that wastewater contains valuable resources, such as nutrients, energy, and water, that be economically recovered (Guest et al., 2009; WEF, 2014; Englehardt et al., 2016). If these resources are recovered in a form that can be sold or re-used, it represents a financial benefit to the public utility responsible for treating the wastewater. In particular, nitrogen (N) and phosphorus (P) are potentially valuable nutrients that are present at relatively high concentrations in wastewater. It has been argued (Macintosh et al., 2018; Jagtap and Boyer, 2018) that recovery of N and P should be performed "upstream" at the building scale or the community scale rather than at the centralized treatment plant, because the

concentrations are progressively diluted as the wastewater moves "downstream" through the sewer system. (Such dilution occurs, for instance, due to water use within the building, groundwater infiltration through leaky pipe connections or failing/degraded pipes, or due to stormwater inflows via legal and illicit stormwater connections into the wastewater pipes). Recovery of nutrients at the building scale should be feasible based on the prevalence of building-scale on-site wastewater treatment systems, which serve approximately 25% of the US population and are used in 30% of new housing developments (Kohler et al., 2017).

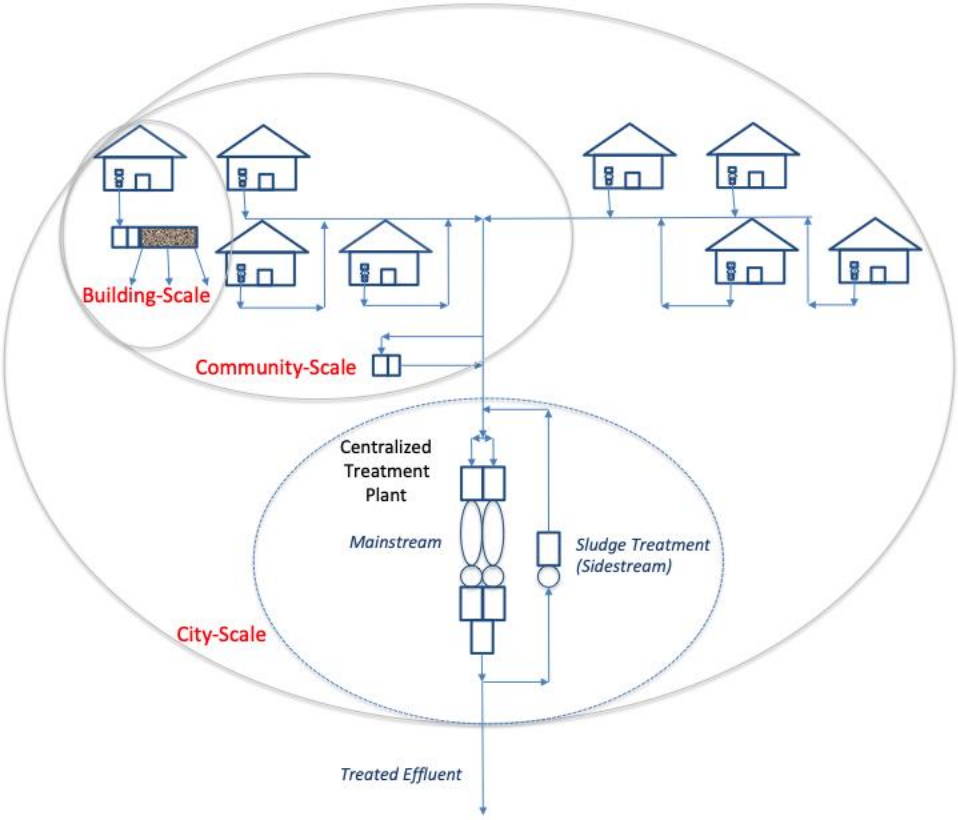

Figure 4. A sewershed can be considered to consist of three scales: building, community, and city scale, the last of which includes both mainstream and sidestream treatment.

Multiple technologies are available for nutrient removal and recovery at the building, community, and city scales (Cornejo et al., 2016). However, selecting from among these available technologies is challenging, especially given the large number of considerations used to evaluate candidate technologies, such as ease of operation and maintenance, value of end products, and cost. To assist in comparing and evaluating candidate technologies, a multi-criteria planning tool (called a House of Quality) was developed that allows stakeholders to "weight" ten technical characteristics according to their priorities (Orner et al., 2017). The planning tool could be used by decision-makers, for example at a municipality or sewershed level, to effectively manage nutrients across scales.

One limitation of the House of Quality planning tool -- or of any existing decision-making tool related to upstream recovery of nutrients -- is that the ramifications of new upstream treatment on existing downstream treatment are unknown. For example, deployment or retrofitting of building-scale and community-scale nutrient removal/recovery technologies where sewers already exist would likely have an effect on the flow rate and/or water quality of the wastewater that reaches the city-scale centralized treatment plant, potentially making the existing treatment infrastructure oversized or less effective. In contrast, building-scale and community-scale technologies could be implemented in new housing developments or new office buildings, which would maintain current hydraulics and water quality at the centralized plant. In recognition of such factors, the state of Iowa has developed five different context categories for managing nutrients from point sources: 1) treatment already installed, 2) treatment not installed and no capacity increases planned, 3) treatment not installed and capacity increases are planned, 4) treatment impractical, and 5) new dischargers (Iowa State University, 2017). Therefore, the impact of upstream nutrient management on centralized treatment is an important unresolved

issue that must be addressed. Furthermore, a similar issue exists within the centralized treatment plant: nutrients can be removed or recovered from "sidestreams" (liquid streams that are recycled within the treatment process), but it is not yet clear how this would affect the mainstream treatment process.

Therefore, the overall goal of this paper is to evaluate how the introduction of new upstream or sidestream nutrient management technologies might affect the treatment efficiency, economics, and environmental aspects of nutrient-management across a sewershed. The sewershed of the Northwest Regional Water Reclamation Facility in Hillsborough County, Florida, is used as a case study. The specific objectives are (1) to develop an "as-is" mass balance for nitrogen and phosphorus for the sewershed; (2) to estimate how the implementation of certain candidate building-, community-, and city-scale nutrient removal and recovery technologies would affect city-scale mainstream treatment in terms of flow rates and loadings of nitrogen and phosphorus; and (3) to make preliminary estimates of the economic benefit to reducing flow rates and nutrient loadings at the centralized treatment plant. These results can inform any community decision-makers on what context-specific nutrient-management technologies to consider at a variety of scales, how these technologies could affect nutrient loading to the wastewater treatment plant, and how these technologies could affect performance and cost of running the wastewater treatment plant.

3.2 Materials and Methods

3.2.1 Site Description

The Northwest Regional Water Reclamation Facility (NWRWRF) is operated by Hillsborough County's Public Utilities department and it serves the region north and west of the City of Tampa wastewater system. NWRWRF is part of the Facilities Accelerating Science &

Technology (FAST) Water Network (http://www.werf.org/lift/LIFT_Test_Bed_Network.aspx), which connects stakeholders like utilities and researchers to accelerate the acceptance of innovative processes and technologies that promote resource recovery (Mihelcic et al., 2017). As a Level 3 facility, NWRWRF provides physical space to pilot innovative water technologies.

This study was done based on historic flow rates of 6.85 mgd (25.9 million L) out of a capacity of 10 mgd (37.9 million L), but the plant is currently undergoing an expansion to 30 mgd (113.6 million L). Current city-scale mainstream treatment includes pretreatment with grit removal & screenings and odor control, 5-stage Bardenpho process, and disinfection. The treated effluent meets permit requirements (annual averages) of 5.0 mg/L biochemical oxygen demand (BOD), 5 mg/L total suspended solids (TSS), 3 mg/L total nitrogen (TN), and 1 mg/L total phosphorus (TP). The sludge produced at NWRWRF is treated through aerobic digestion at an adjoining biosolids management facility; a liquid sidestream returns from sludge processing to the headworks of the mainstream treatment train.

3.2.2 Estimating Baseline ("as-is") Nutrient Loadings

Hillsborough County water quality data are provided in Table 8 for five locations across the NWRWRF sewershed. Two of the locations, Henderson and Tudor Chase, are pump stations that collect wastewater from 297 and 700 homes, respectively (i.e. exiting community-scale). The NWRWRF influent concentrations of 35.9 mg/L of ammonia and 12.9 mg/L of total phosphorus (TP) are close to values that Metcalf and Eddy (2014) considers to be highly loaded (41 mg/L of ammonia, 11 mg/L of TP). Based on the average flow rate of 18 m^3/min and the reported concentrations of TKN and TP in the influent to NWRWRF, the estimated loadings to NWRWRF are 1370 kg/d TKN and 335 kg/d TP. Of those totals, the estimated loadings due to sidestream recycle are 148 kg/d TN (11% of total) and 187 kg/d TP (56% of total). The nutrient

loadings in the NWRWRF treated effluent are 59 kg/d TN (96% removal from influent) and 11 kg/d TP (97% removal from influent).

Table 8. Water quality and hydraulic data at five locations in the sewershed. Data provided by Hillsborough County.

	Henderson	Tudor Chase	WWTP Influent	WWTP Sidestream	WWTP Effluent
# Houses	297	700			
# Pump Stations	1	4	75		
Flow Rate (mgd)			6.85	0.29	6.85
TN (mg/L)	54.7	39.8		135	2.29
TKN (mg/L)	54.7	39.8	52.7		1.1
NH_3-N (mg/L)	44.4	34.7	35.9		0.16
TP (mg/L)	6.7	5.5	12.9	170	0.42
BOD (mg/L)	250	263	259	267	0.7

3.2.3 Candidate Technologies for Nutrient Management

A literature review was conducted to identify 40 nutrient-management technologies; of these, 14 could be deployed at the building scale, 17 could be deployed at the community scale, and 9 could be deployed in the sidestream. To determine which of these 40 technologies were worth considering in more detail, the House of Quality planning matrix was used. The House of Quality ranks technologies at multiple scales based on ten technical characteristics (Orner et al., 2017). The weightings for the ten technical characteristics, shown in Table 9, were provided by representatives from Hillsborough County. These weightings, in conjunction with the technologies and scoring developed in the planning matrix, provided recommended nutrient removal and recovery technologies at the building scale, community scale, and sidestream to be further evaluated to determine their likely impacts on flow rates, nutrient loadings, and cost. The resultant Houses of Quality are provided in the Appendix in Tables B1–B3.

Table 9. Weightings for ten technical characteristics for building-scale, community-scale, and city-scale nutrient removal and recovery technologies. The weightings were provided by representatives of the Northwest Regional Water Reclamation Facility operated by Hillsborough County, Florida.

	Technical Characteristic	Abbreviation	General Description	Weight
1	Ease of operation and maintenance	O&M	The lack of complexity and resources needed in the technical handling of the technology after it has been installed	1.2
2	Size/footprint	SF	Extent to which the technology requires physical space	1.2
3	Capital cost	CC	Cost to install and ready the technology for use, as compared to the 'baseline' technology of sewer connections with centralized treatment	1.0
4	Operational cost	OC	Extent to which the product requires operational input in terms of energy, chemicals, water, and labor	1.0
5	Value of end products	EP	Extent to which the technology provides valuable products, such as nutrients, energy, water, and commodities	0.5
6	Technical maturity	TM	Readiness of a technology to be effectively used in nutrient removal or recovery as reported in literature	0.9
7	Environmental impact	EI	Extent of the impact to the environment on the basis of air pollution, land pollution, release of nitrogen, and release of phosphorus	1.1
8	Performance in nitrogen removal	PN	Extent of nitrogen removal or recovery by the technology	1.1
9	Performance in phosphorus removal	PP	Extent of phosphorus removal or recovery by the technology	1.1
10	Aesthetics	A	Extent to which the technology is acceptable to stakeholders on the basis of visual appeal, noise level, and odors	0.9

3.2.4 Estimating Effects of Upstream/Sidestream Technologies on Mainstream Flow Rates and Nutrient Loadings

The technologies recommended by the House of Quality tool could potentially reduce the volumetric flow, nitrogen loading, and/or phosphorus loading that reaches the city-scale mainstream treatment at NWRWRF. Expected reductions (on a percentage basis) of flow rate, nitrogen loading, and phosphorus loading to NWRWRF were estimated by reviewing literature for each technology selected by the House of Quality. The current flow rate (L/d), TN loading (g/d), and TP loading (g/d) were multiplied by the estimated reduction efficiency (%) to determine the diverted quantities.

For example, composting toilets are estimated to reduce the flow rate exiting a building by 31% and reduce the TN loading by 90% (Table 3). Therefore, composting toilets would divert 408 L/d (109 gpd) of the original 1320 L/d (350 gpd) in a building in the Henderson community, meaning that 912 L/d (241 gpd) would exit the building. Also, toilets would divert all but 7.2 g/d TN of the original 72.2 g/d TN from the building. (For the purposes of this analysis, it is assumed that the nutrients in the compost would be safely recovered and utilized.) Supposing that all houses in the sewershed installed composting toilets, and the wastewater treatment plant (WWTP) only received flow from houses, the flow entering the WWTP would be reduced by 31% (808,000 L/d) of the as-is 25,900,000 L/d (6.85 mgd). The daily as-is TN mass load entering the WWTP is 1,360 kg/d, calculated by multiplying the as-is flow (25,900,000 L/d) by the as-is TKN concentration (52.7 mg/L TN). If all houses installed composting toilets, 90% of the TN would be diverted (1224 kg/d). Therefore, the daily estimated TN mass load (kg/d) entering the mainstream plant would only be 136 kg/d.

3.2.5 Estimate of Potential Cost Savings

Costs can be avoided by implementing nutrient management technologies at the building-, community-, and city-scale (sidestream) to reduce nutrient loading entering the treatment plant. For example, a reduction in nitrogen loading could reduce aeration costs needed for nitrification by \$0.61/kg NH_3-N and methanol costs needed for denitrification (at treatment plants that add methanol as an external carbon source and electron donor) by \$6.38/kg NH_3-N (Drexler et al., 2014). A reduction in phosphorus could reduce mainstream costs for alum by \$0.61/kg P (Cunningham et al., 2018). Costs associated with implementing the technologies are not included in this study.

3.3 Results and Discussion

3.3.1 Technology Prioritization at Building-, Community-, and City-Scale (Sidestream)

Results from the building-scale House of Quality, shown in Appendix B1, indicate that the top-scoring technologies are composting toilets (31.8) and direct urine application as fertilizer (27.1). Composting toilets placed slightly ahead of conventional wastewater treatment (flush toilet connected to a sewer) (29.7), and both technologies placed well ahead of septic systems (24.1), which received low marks in value of end products, environmental impact, and performance in nitrogen and phosphorus removal. The two top technologies scored better than conventional wastewater treatment in their reduced environmental impact and their ability to recover nitrogen and phosphorus, although these technologies require more maintenance than conventional treatment, require more space in each building, and are slightly less technically mature than conventional treatment.

Results from the community-scale House of Quality, shown in Appendix B2, indicate that the top scoring technologies are constructed wetlands (37.1) and biological nutrient removal

(defined as nitrification and denitrification with enhanced biological phosphorus removal) (31.2). While constructed wetlands scored well because of their low operation and maintenance requirements and their low capital and operating costs, biological nutrient removal benefitted from high scores in removal of nitrogen and phosphorus.

Results from the city-scale sidestream House of Quality, shown in Appendix B3, indicate that the two technologies that scored higher than the default scenario (no sidestream treatment) are struvite precipitation (35.3) and ion exchange (31.3). However, NWRWRF digests sludge aerobically, not anaerobically. Therefore, technologies such as struvite precipitation that require anaerobic digestion are not considered for this case. Ion exchange scored better than a no-treatment strategy in its ability to produce valuable end products and to recover nitrogen and phosphorus, although it requires more space, capital costs, and operational costs.

3.3.2 Estimated Reductions from Building-Scale Treatment Technologies

The estimated reduction percentages of flow rate, nitrogen loading, and phosphorus loading of the two building-scale technologies are shown in Table 10. Source separation of urine would not have a large effect on the flow to the WWTP. However, composting toilets are a dry toilet technology (e.g. do not use flushwater) and would have a 31% reduction on the flow. Because composting toilets and source separation each divert urine, the nitrogen and phosphorus loading would be greatly reduced.

3.3.3 Estimated Reductions from Community-Scale Treatment Technologies

The estimated reduction percentages of flow rate, nitrogen loading, and phosphorus loading of the two community-scale technologies are shown in Table 11. Constructed wetlands at the community scale would reduce flow by 30%, whereas biological nutrient removal would not

reduce flow. However, both community-scale technologies would reduce total nitrogen and total phosphorus concentrations by at least 60%.

Table 10. Building-scale technologies. Total Nitrogen (TN) and Total Phosphorus (TP) loads were calculated multiplying building flow by measured TN and TP concentrations at pump stations (54.7 mg/L TN and 6.7 mg/L TP, respectively).

	As-Is		Composting Toilets			Direct Urine Application as Fertilizer		
	Henderson	Tudor Chase	Reduction (%)[1]	Henderson Output to Sewer per Building	Tudor Chase Output to Sewer per Building	Reduction (%)[2]	Henderson Output to Sewer	Tudor Chase Output to Sewer
Building Flow (L/d)	1320	1700	31 +/- 11	912	1170	1	1310	1690
TN Load (g/d)	72.2	93.0	90 +/- 0	7.2	9.3	82 +/- 8	13.3	17.1
TP Load (g/d)	8.84	11.4	86 +/- 6	1.28	1.65	58 +/- 11	3.69	4.75

[1] Gerba et al., 1995; Jonsson et al., 2000; de-Bashan & Bashan, 2004; Hargreaves & Warman, 2007; Jamrah et al., 2008; Hotta & Funamizu, 2008; Tilley et al., 2014; Orner et al., 2018

[2] Larsen et al., 1996; Hanaeus et al., 1997; Jonsson et al., 2000; Karak & Bhattacharyya, 2011; Mihelcic et al., 2011; Landry & Boyer, 2016

Table 11. Community-scale technologies. Total Nitrogen (TN) and Total Phosphorus (TP) loads were calculated multiplying community flow by measured TN and TP concentrations at pump stations (54.7 mg/L TN and 6.7 mg/L TP, respectively).

	As-Is		Constructed Wetlands			Biological Nutrient Removal		
	Henderson	Tudor Chase	Reduction (%)[3]	Henderson Output to Sewer	Tudor Chase Output to Sewer	Reduction (%)[4]	Henderson Output to Sewer	Tudor Chase Output to Sewer
Community Flow (L/d)	379000	1210000	30 +/- 28	265000	848000	0	379000	1210000
TN Load (kg/d)	20.7	66.2	66 +/- 18	7.08	22.6	91 +/- 8	1.93	6.17
TP Load (kg/d)	2.54	8.11	60 +/- 17	1.02	3.26	82 +/- 14	0.46	1.46

[3] Tanner et al., 1995; Tanner et al., 1998; Brown et al., 2000; Woltemade, 2000; Comas, 2004; Tchobongolous et al., 2004; Gross et al., 2007; Ye & Li, 2009; Kadlec, 2010; Ortega et al., 2011; National Small Flows Clearinghouse, 2019

[4]Obaja et al., 2003 Obaja et al., 2005; Ersu et al., 2010; Monclus et al., 2010; Molinos-Selnante et al., 2012; Iowa State University, 2017

3.3.4 Estimated Reductions from City-Scale (Sidestream) Treatment Technologies

The estimated reduction percentages for flow rate, nitrogen loading, and phosphorus loading for the city-scale sidestream technology are shown in Table 12. While ion exchange does not reduce flow, it reduces total phosphorus loading by 79% and total nitrogen loading by 72%. WWTPs that utilize aerobic digestion, such as NWRWRF in Hillsborough County, FL, will not be able to implement technologies such as struvite precipitation that require anaerobic digestion.

Table 12. Sidestream impacts on mainstream treatment at wastewater treatment plant. Total Nitrogen (TN) and Total Phosphorus (TP) loads were calculated multiplying sidestream flow by measured TN and TP concentrations in the sidestream (135 mg/L TN and 170 mg/L TP, respectively).

	As-Is	Ion Exchange Reduction (%)[5]	Ion Exchange Output to Mainstream
Flow (L/d)	1100000	0	290000
TN Load (kg/d)	149	79 +/- 14	28
TP Load (kg/d)	187	72 +/- 13	47

[1] Zurita et al., 2006; Thornton, 2007; Martin et al., 2009; Seo et al., 2013; Kim et al., 2014

3.3.5 Impact of Building-, Community-, and City-Scale Sidestream Treatment Technologies on Mainstream Treatment

While 1,360 kg/d of total nitrogen arrives to the plant from the sewershed, less than 200 kg/d arrives from the sidestream (Figure 5). Thus, building-scale source-separation technologies (i.e. composting toilets and direct urine application as fertilizer) and community-scale technologies all reduce total nitrogen loads by over 800 kg/d. More phosphorus loadings come from the sidestream (187 kg/d) than the sewershed (174 kg/d). Thus, the sidestream technology outperforms the previously mentioned source-separation and community-scale technologies in reducing phosphorus loading.

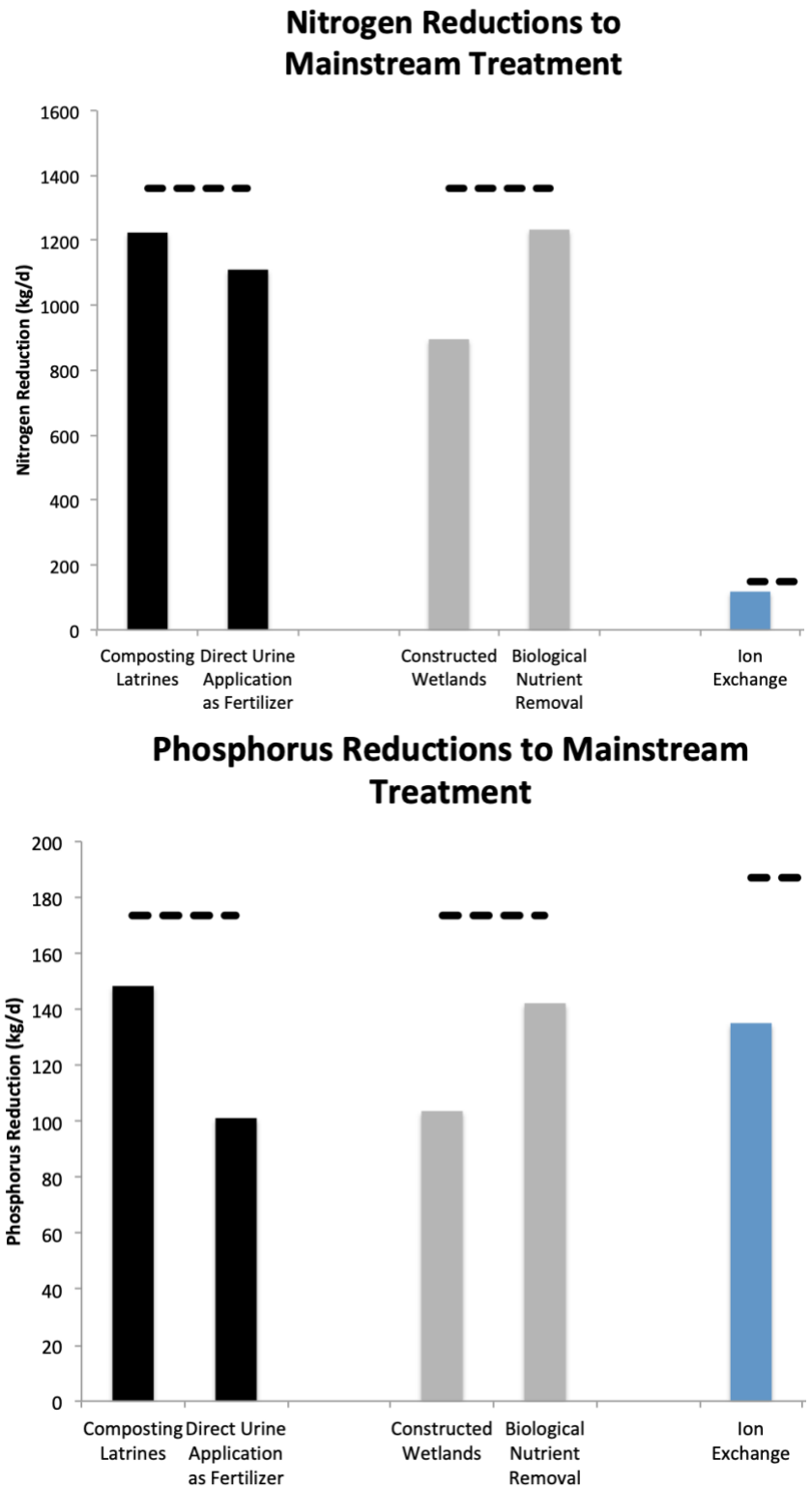

Figure 5. Nitrogen (above) and phosphorus (below) reductions to mainstream treatment. Black are building-scale technologies, grey are community-scale technologies, and dark blue are city-scale sidestream technologies. Dashed lines are nutrient mass loadings entering each technology.

3.3.6 Potential Cost Savings from Building-, Community-, and City-Scale Technologies on Mainstream Treatment

Potential cost savings from implementing each technology are shown in Table 13—the savings range from $90 to $600 per day. Additionally, if treatment technologies are implemented where treatment didn't previously exist, shadow prices (estimated price of a service for which no market price currently exists, such as environmental benefits) could also be considered. For example, if nitrogen and phosphorus were previously discharged to a river, shadow prices could be considered at $19/kg N treated and $36/kg P treated (Hernandez-Sancho et al., 2010).

These potential cost savings must be balanced against the cost of implementing the candidate technologies. At the building scale, implementing composing toilets would require purchase and installation of new toilets and operation and maintenance of the compost. Likewise, urine separation would require installation of new source-separation toilets and maintenance of collecting and applying the urine end product. Detailed cost estimation is beyond the scope of this paper, but there are models available that could provide information for a more detailed cost-benefit analysis. At least two models of composting toilets exist that mimic pour-flush toilet designs. For example, Clivus Multrum (2018) produces a foam-flush toilet and also handles maintenance tasks such as adding bulking material, monitoring moisture content, and disposing of liquid and solid end products. Terra Preta is another composting toilet model that handles operation and maintenance tasks (Factura et al., 2010).

Similar cost considerations must be made for community-scale and sidestream technologies. At the community scale, constructed wetlands would have high installation costs of purchasing land and excavation, but then the system is passive with minimal maintenance. Conversely, biological nutrient removal would require both installation and maintenance costs.

In the sidestream, ion exchange would require installation and maintenance costs. However, maintenance costs may be minimal because the reactors are on-site at the treatment plant and could reduce maintenance to the mainstream treatment process.

The cost savings presented in this study assumed 100% of buildings or communities adopted each technology, producing the maximum reduction. If only 10% or 50% adoption was achieved, both the cost savings from nutrient loading reduction and implementation costs would also drop accordingly.

3.3.7 Application to Sewersheds other than NWRWRF

The results of nutrient reduction and cost savings from the NWRWRF sewershed can be applied to other urban and suburban sewersheds. The findings that 1) building- and community-scale technologies can reduce nitrogen loading by over 50%, and 2) sidestream treatment has the potential to be effective in reducing phosphorus loading, are generally applicable. In contrast, the specific technologies recommended are site specific. However, the House of Quality multi-criteria planning tool contains weightings that can be modified by stakeholders to easily select technologies in other sewersheds.

Table 13. Cost savings per day by implementing building-, community-, and city-scale sidestream technologies.

	Building- Scale Technologies				*Community- Scale Technologies*			
	Composting Toilets		**Direct Urine Application as Fertilizer**		**Constructed Wetlands**		**Biological Nutrient Removal**	
	Reduced Quantity	**Total Savings**	**Reduced Quantity**	**Total Savings**	**Reduced Quantity**	**Total Savings**	**Reduced Quantity**	**Total Savings**
Nitrification	834	$509	756	$461	610	$372	840	$512
Alum	148	$23	101	$16	104	$16	142	$22
Total Savings		$532		$477		$388		$535

				Sidestream Ion Exchange	
	Unit Savings	**Unit**	**Original Quantity**	**Reduced Quantity**	**Reduced Quantity**
Nitrification	$0.61[1]	per kg NH_3-N	148	118	$72
Alum	$0.16[2]	per kg P	187	135	$21
Total Savings					$93

[1]from Drexler, 2014
[2]from Cunningham et al., 2018.
Calculated from the following: ($0.04 / lb) * (1390 lb / d) / (361 kg P / d)

3.4 Conclusion

The introduction of new upstream nutrient management technologies affects the treatment efficiency and economics of nutrient management across a sewershed. A number of conclusions can be made to inform decision-makers what context-specific nutrient management technologies to consider at a variety of scales and how these technologies could affect nutrient loading to the wastewater treatment plant. Plans to reduce nitrogen loading through implementing nutrient-management technologies should be considered at a variety of scales, including the building scale and community scale. Results from this case study indicate that building-scale source separation and community-scale nutrient management could reduce nitrogen loading to the mainstream treatment train of the centralized wastewater treatment plant by over 50%, but sidestream treatment has minimal impact in nitrogen reduction. Conversely, sidestream treatment technologies such as ion exchange are the most effective in reducing phosphorus loading to the mainstream due to high quantities of phosphorus recycling back to the head of the plant; building- and community-scale technologies are expected to be only moderately effective in reducing phosphorus loading to the treatment plant.

In addition to the cost considerations discussed above, public utilities might wish to consider other factors such as: 1) Is the treatment plant near capacity? 2) Is population growth expected? 3) Are regulations favorable to utilizing nutrient products (e.g. soil conditioner)? 4) Are community members or farmers interested in applying nutrient products on their soil? If answers to the above considerations are favorable, nutrient-management infrastructure could be initially piloted in buildings or neighborhoods like Henderson and Tudor Chase.

These results can inform decision-makers about what context-specific nutrient management technologies to consider at a variety of scales. In this case, the analysis shows that

Hillsborough County could consider sidestream treatment technologies like ion exchange to reduce phosphorus loading in the mainstream. Additionally, the County might wish to consider a pilot project of building-scale or community-scale infrastructure. A favorable opportunity for the pilot project could be a new development that could install new toilets for composting or urine separation, or that has land available for constructed wetlands. Overall, upstream and sidestream technologies can be deployed to reduce nitrogen and phosphorus loading at centralized wastewater treatment plants and provide daily cost savings of up to $600 per day.

CHAPTER 4: ENERGY RECOVERY AND NITROGEN MANAGEMENT FROM STRUVITE PRECIPITATION EFFLUENT VIA MICROBIAL FUEL CELLS[6]

4.1 Introduction

Wastewater treatment now integrates recovery of resources including nutrients, energy, and water (Guest et al. 2009; Latimer et al. 2015; Englehardt et al. 2016; WE&RF 2017). An established technology that recovers nitrogen and phosphorus from waste streams is the precipitation of struvite ($MgNH_4PO_4$), which may be commercialized as a fertilizer (Battistoni et al. 2000; Le Corre et al. 2009; de-Bashan and Bashan 2004; Yetilmezsov et al. 2017). As of 2013, struvite was approved for use as a fertilizer in Canada, the European Union, and 34 states in the U.S. (Cullen et al. 2013). Engineered struvite precipitation (ESP) can be applied to wastewater treatment plant sidestreams (Mehta et al. 2014; Milbrandt 2005), urine (Lind et al. 2000; Barbosa et al. 2016), landfill leachate (Gunay et al. 2008; Huang et al. 2014) and agricultural waste (Song et al. 2011; Amini et al. 2017). Several companies are installing full-scale struvite precipitation systems around the world at wastewater treatment plants, including Ostara Nutrient Recovery Technologies, Inc., which has installed at least 17 Pearl systems serving 11.5 million people (Ostara 2018).

Depending on the source water, the liquid effluent from ESP is likely to still contain either nitrogen (N) or phosphorus (P) if one of them is present in stoichiometric excess. When

[6] Reprinted with permission from ASCE: Orner, K.D., Cools, C., Zalivina, N., Balaguer-Barbosa, M., Mihelcic, J.R., Chen, G., and Cunningham, J.A. (2019) "Energy recovery and nitrogen management from struvite precipitation effluent via microbial fuel cells," *Journal of Environmental Engineering*, 145(3): 04018145. Permission is included in Appendix A.

performed on the sidestream of a wastewater treatment plant (i.e., the liquid effluent of a sludge digester that is recycled back to the plant headworks), ESP typically recovers 80-90% of P but only ~20% of N (Mehta et al. 2014). This is because sidestreams usually contain molar concentrations of N that are considerably higher than those of P. The nitrogen remaining in the liquid effluent from ESP, approximately 700–900 mg/L as N (Mehta et al. 2014), is typically recycled back to the mainstream biological treatment process. This can cause problematic nutrient load variations and decreased overall nitrogen removal efficiency. It may also increase the cost of wastewater treatment because of additional aeration (for nitrification) and chemical costs (for denitrification, if external electron donors are required) (Wett and Alex 2003).

To avoid increased nitrogen loading in the mainstream, technologies to recover or remove nitrogen from the sidestream need to be deployed after ESP. If cost-effective, processes that recover nitrogen in a useable form -- e.g., ammonia stripping or microbial electrochemical systems (Gustin and Marinsek-Logar 2011; Zhang and Angelidaki 2015a; Zhang and Angelidaki 2015b) -- are preferable to processes that simply remove nitrogen from the system. However, recovery processes often require the input of energy or chemicals and have often been found to be too expensive (Eekert et al. 2012). Biological removal processes such as conventional nitrification/denitrification, single reactor system for high-activity ammonium removal over nitrite (SHARON), completely autotrophic nitrogen removal over nitrite (CANON), and/or anaerobic ammonium oxidation (ANAMMOX) may therefore be deployed to remove nitrogen from the effluent of ESP. However, these technologies also require a net input of energy. Ideally, a technology deployed in conjunction with ESP would remove or recover excess nitrogen without requiring additional input of chemicals or energy.

A microbial fuel cell (MFC) may be an appropriate technology to generate energy from the liquid effluent of ESP while further removing nitrogen. In the anodic chambers of MFCs, organic compounds are oxidized to release electrons, which are transferred exogenously to the anode (Logan et al. 2006). Through the circuit, the electrons are transferred to the cathode, released, and consumed by electron acceptors. In the proposed MFC, the electron acceptor could be nitrate or nitrite, which would be converted to nitrogen gas and removed. This proposed technology differs from the aforementioned BNR technologies because it would recover energy (in the form of electricity) in addition to removing nitrogen. The proposed MFC has a technology readiness level (NASA 2018) of 4 out of 9 as the process has been validated in the laboratory environment.

In a similar technology with synthetic influents, Clauwaert et al. (2007) achieved complete denitrification of NO_3^- in the cathodic chamber of an MFC. Virdis and co-workers built on the work of Clauwaert et al. (2007) to further demonstrate using synthetic influents that MFC systems could simultaneously remove carbon and nitrogen from wastewater while generating electricity (Virdis et al. 2008, 2009, 2010). Additionally, MFCs with a separate nitrification stage have only been developed to remove nitrogen and obtain power from two studies with real waste streams--landfill leachate (Lee et al. 2013) and the liquid effluent of a latrine (Castro et al. 2014). However, to the best of our knowledge, MFCs have not yet been applied for the removal of nitrogen from the liquid effluent of ESP to remove N and recover energy. The chemistry and biology of ESP effluent may differ considerably from that of waste streams previously treated by MFCs for nitrogen removal in terms of total nitrogen (TN), chemical oxygen demand (COD), and pH; for instance, whereas the liquid effluent of a latrine contained 100 mg TN/L and pH of 7, and diluted landfill leachate contained 120 mg TN/L, 266

mg COD/L, and pH of 8.4, the effluent from ESP can exceed 1500 mg TN/L, 4000 mg COD/L, and pH of 8.5. These differences need further research because previous research has shown that the strength of wastewater and diversity of microbial community can affect power production (Pant et al. 2010; Castro et al. 2014).

Therefore, the overall objective of this chapter is to evaluate the performance of a proof-of-concept process that includes a fixed-film nitrification reactor and two-chambered MFC for energy generation and nitrogen removal from the liquid effluent of ESP. Specifically, we will (1) quantify the N removal achieved by applying a fixed-film nitrification reactor and MFC downstream of ESP, (2) quantify the power generated by the MFC, and (3) evaluate the energy generation and nitrogen removal performance to determine if the proof-of-concept process merits further consideration in treating the liquid effluent of ESP. The long-term impact of this research will be transform wastewater treatment plants into resource recovery facilities of the future by supporting technological innovation that recovers multiple resources (Mihelcic et al. 2017). Furthermore, if successful, this technology will offset operational costs through nutrient removal and recovery and energy production that can be used to power locally situated equipment, representing cost savings for wastewater treatment plants.

4.2 Materials and Methods

4.2.1 Process Overview

The process to be evaluated, shown in Figure 6 includes a fixed-film nitrification reactor and a two-chambered microbial fuel cell. The fixed-film nitrification reactor treats the liquid effluent from ESP (stream 1 in Figure 6) by supplying oxygen to promote the conversion of ammonium to nitrite or nitrate. The fixed-film nitrification effluent (stream 2) is the influent to the cathodic chamber of the microbial fuel cell (location 3), where the anaerobic environment

promotes denitrification. The anodic chamber of the microbial fuel cell (location 4) uses carbon from filtered raw wastewater (stream 5) as an electron donor for organic decomposition and produces protons and electrons that are utilized in the cathodic chamber.

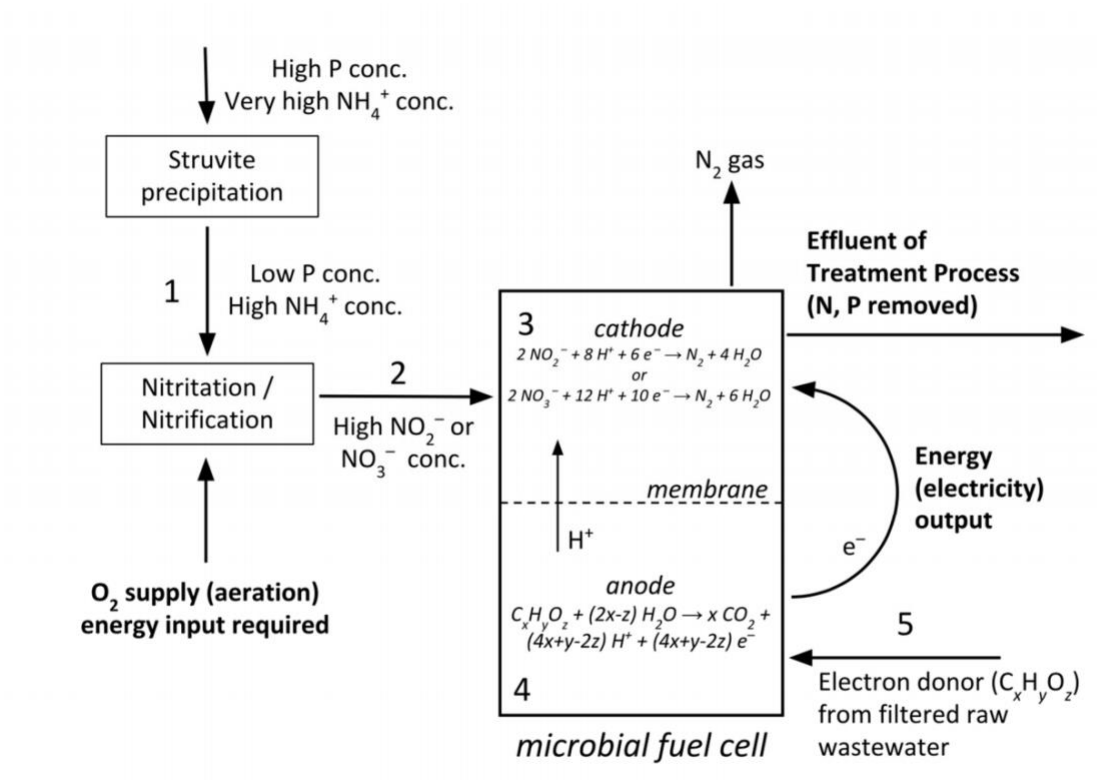

Figure 6. Schematic of proposed treatment and recovery process. The numbers indicate the five different sampling locations in the laboratory-scale system.

4.2.2 Engineered Struvite Precipitation (ESP)

To test the ability of the MFC to treat liquid effluent from ESP, the effluent from a struvite precipitation process was used. Influent to the struvite precipitation process (conducted by Maraida Balaguer-Barbosa) was centrate (i.e., centrifuged effluent) from a lab-scale anaerobic digester (operated by Nadezhda Zalivina), which in turn treated waste activated sludge from a nearby wastewater treatment plant. The struvite precipitation process was performed in batches

in a 3.5-L reactor. Centrate from the anaerobic digester was amended with $MgCl_2 \cdot 6H_2O$, the pH was adjusted to 8.5 via addition of NaOH, and struvite seed crystals were added to aid in nucleation. Following chemical addition to the sidestream, the reactor was operated at a mixing speed of approximately 150 rpm for 8-10 minutes to allow precipitation to occur. Solids were then separated from the liquid phase via centrifugation at 5000 rpm for 20 min.

The ESP process recovered, on average, approximately 16% of the ammonium and 73% of the phosphate from the digester centrate. The ammonium removal percentage was in the range of typical commercial options available to recover struvite; the phosphate recovery percentage was slightly below the typical observed range of ~80–95% (WEF 2014). Because an average of 84% of the influent ammonium remained in the liquid effluent of ESP reactor, this liquid was an appropriate feedstock to test the concept of nitrification and MFC for simultaneous energy recovery and nitrogen removal.

4.2.3 Fixed-Film Nitrification Reactor

The liquid effluent from ESP was fed into the fixed-film nitrification reactor, where it was aerated with a fish-tank aerator to promote nitrification. The average concentration of dissolved oxygen (DO) in the nitrification reactor was 6.8 mg/L. The volume of the fixed-film nitrification reactor was 0.4 L and a total of 0.07 L was removed over three sampling events each week to provide influent for the cathodic chamber and samples for analysis. Therefore, the average hydraulic residence time (HRT) in the fixed film nitrification reactor was 5.8 weeks. Although a HRT of this duration would not be realistic for full-scale operation, it was appropriate for this proof-of-concept study because it provided the necessary volume of nitrified feed to the MFC. To prevent wash-out of the nitrifying bacteria, hollow plastic 1-cm diameter

carriers (Wholesale Koi Farm, Norco, CA) were placed in the reactor to provide additional surfaces for biofilm growth (Gong et al. 2011).

4.2.4 Microbial Fuel Cell

The MFC consisted of two chambers, each of which was a glass reactor with a liquid volume of 100 mL, joined by a glass bridge with a CMI-7000 cation exchange membrane (Membranes International Inc., Ringwood, NJ). Although the two-chamber design is not state-of-the-art for MFC applications, it was appropriate for this proof of concept because electron-donating and electron-accepting processes were separated into different chambers where they could be monitored and evaluated. The influent to the anodic chamber was raw wastewater filtered by a vacuum pump (stream 5 in 6), and the influent to the cathodic chamber was the liquid effluent of the fixed-film nitrification reactor (stream 2 in Figure 6). The electrodes (anode and cathode) inside the chambers were constructed of 0.5 mg/cm^2 60% platinum on Vulcan-Carbon Paper (Fuel Cell Store, College Station, TX) with a surface area of 6.45 cm^2 for each electrode. Anoxic conditions were maintained in both the anodic and cathodic chambers via a gentle purge with nitrogen gas. The pH and concentration of dissolved oxygen were measured throughout the system as described in Table 14.

Table 14. Sampling locations and analyses conducted for assessment of the system performance.

Analyte	Method	Sampling Locations in Figure 6
Total Nitrogen	APHA 2012 Standard Method 4500-N (Persulfate)	1,2,3
Total Phosphorus	APHA 2012 Standard Method 4500-P E	1,2
Anions (NO_2^-, NO_3^- and PO_4^{3-})	APHA 2012 Standard Method 4110B	1,2,3
Cations (NH_4^+, Mg^{2+}, Ca^{2+})	ISO 14911 (ion chromatography)	1,2,3,4,5
Alkalinity	APHA 2012 Standard Method 8221	1,2,3,4,5
Dissolved Oxygen	Thermo Scientific Orion (Waltham, MA)	1,2,3,4,5
pH	Thermo Scientific Orion (Waltham, MA)	1,2,3,4,5
Chemical Oxygen Demand	APHA Standard Method 5220B	4,5

The anode of the MFC was inoculated with *Shewanella putrefaciens*, and the cathode was inoculated with *Geobacter metallireducens*, both obtained from the American Type Culture Collection (ATCC) (Manassas, VA). Both bacteria were cultivated in broth as described on the ATCC website (ATCC 2018). During a start-up period, the wastewater sources used as influents for the MFC were artificial solutions of glucose (280 mg/L, used as carbon source for organic decomposition) for the anodic chamber and sodium nitrate (340 mg/L, used as nitrate source for denitrification) for the cathodic chamber. Once the MFC was stabilized after about 28 days, the anodic chamber influent was transitioned to filtered raw wastewater from Northwest Regional Water Reclamation Facility (Hillsborough County, FL) and the cathodic chamber influent was transitioned to effluent of the fixed-film nitrification reactor, as shown in Figure 6.

Voltage and current in the MFC were measured with a Keithley 2701 digital multimeter (Solon, OH) in closed-circuit mode. A 1000-□ resistor was placed in the circuit between the anode and cathode to provide a load (external resistance). The external resistance of 1000 □ was

chosen because it generated the greatest power output of the MFC. The selected external resistance was consistent with the estimated internal resistance of 2027 □, estimated via the current interrupt method (Aelterman et al. 2006).

The MFC was operated for 201 days. Anodic effluent, cathodic effluent, and fixed-film nitrification effluent were removed for sampling and replaced with the appropriate feed streams. If the liquid volume in the anodic or cathodic chambers remained below 100 mL after replacement (if liquid volume was lost, for example, due to evaporation), deionized water was added to maintain a constant reactor volume.

4.2.5 Sampling and Analysis

Details of analytes measured, analytical methods used, and the analyses done at each sample location are shown in Table 13. Anion and cation analyses were performed using a Metrohm Peak 881 AnCat (Herissau, Switzerland) ion chromatography (IC) system.

4.3 Results and Discussion

4.3.1 Nitrogen Fate in Fixed-Film Nitrification Reactor

The fixed-film nitrification reactor decreased the TN concentration by an average of 37%, from 1530 ± 130 mg N/L in the struvite effluent to 960 ± 150 mg N/L in the fixed-film nitrification reactor effluent during the 28-w operation period (Figure 7). This observation was somewhat surprising because the conversion of ammonium to nitrate using aeration to promote nitrification would not be expected to decrease the total nitrogen concentration. A possible explanation is that simultaneous nitrification and denitrification occurred in the fixed-film nitrification reactor, perhaps due to the formation of biofilm on the carriers and the consequent localized gradients of the concentration of dissolved oxygen (Masuda et al. 1991; Virdis et al.

2010)[7]. Another explanation for total nitrogen reduction is that biological assimilation could be occurring due to the growth of nitrifying bacteria during the reactor's long residence time[8].

The fixed-film nitrification effluent (stream 2 in Figure 5) had the following nitrogen concentrations: 220 ± 70 mg/L NH_4^+-N, 360 ± 70 mg/L NO_2^--N, and 14 ± 12 mg/L NO_3^--N. Therefore, the total inorganic nitrogen was about 590 ± 150 mg/L, which can be compared to the observed total nitrogen of 960 mg/L; the difference of 370 mg/L is presumed to be organic nitrogen present in the microbial biomass.

The results from the fixed-film nitrification reactor indicated that ammonium was primarily being converted into nitrite, not nitrate. As has been seen in other studies, nitrite-oxidizing bacteria are suppressed by high concentrations of free ammonia or by low concentrations of DO (Anthonisen et al. 1976; Kouba et al. 2014). Based on the average measured concentration of NH_4^+-N of 220 mg/L, temperature of 23 °C, pKa value of 9.24 (Brown et al. 2011), and pH of 6.0, it was estimated that the free ammonia concentration in the fixed-film nitrification effluent was 0.14 mg/L NH_3-N, which is in the range that has been shown to suppress nitrite-oxidizing bacteria (Anthonisen et al. 1976; Kouba et al. 2014).

4.3.2 Nitrogen Removal in Microbial Fuel Cell

Although the fixed-film nitrification reactor was originally intended as a nitrification reactor, and I still refer to it as a nitrification reactor, in fact the dominant process was nitritation rather than nitrification. The fixed-film nitrification reactor effluent, which included more nitrite than nitrate, was used as influent to the cathodic chamber of the MFC (see Figure 5). In the

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[7] Nitrogen reduction due to ammonia volatilization is neglible. Ammonia volatilization is higher when temperatures are above 30 °C, and in this case temperature was 23 °C. Likewise, the pH of 6.0 of the nitrification effluent is well below the NH_3 pKa of 9.24.

[8] Nitrogen reduction due to assimilation into microbial biomass is not negligible.
mu_nit = mu_max_nit * S_NH4 / (k_sat_NH4 + S_NH4).
Given that mu_max_nit = 0.5 (1/d), k_sat_NH4 = 1.1 (mg NH4/L), S_NH4 = 1380 (mg NH4/L), then mu_nit = 0.5 (1/d). Assuming that mu_death = 0.1 (1/d) and X = 40 (mg biomass/L),
N_red = 0.4 (1/d) * 5.8 w * 7 d/w * 1 mg N/8 mg biomass = 80 mg N/L

cathodic chamber, the average TN decreased by about 24%, from an average input concentration of 960 mg/L to approximately 730 ± 110 mg/L N. The cathodic effluent had concentrations of 150 ± 40 mg/L NH_4^+-N, 50 ± 20 mg/L NO_2^--N, and 2 ± 2 mg/L NO_3^--N over the 28-week operation. The drop in nitrite concentration from 360 mg/L N to 50 mg/L N indicated that the cathodic chamber primarily utilized denitritation to remove nitrogen. The nitrogen data are shown graphically in Figures 6 and 7.

The overall nitrogen removal achieved by the treatment process (i.e., from the struvite effluent to the MFC cathode effluent) was approximately 52% over the 28-week period of operation.[9]

[9] Factors affecting nitrogen removal in microbial fuel cells utilizing autotrophic cathodic denitrification include dissolved oxygen (DO), pH, and carbon/nitrogen (C/N) ratio (Kelly and He, 2014). Nitrification in the external nitrification chamber could be improved by increasing DO levels or by increasing residence time (Yan et al., 2016). Denitrification could be improved by decreasing DO levels in the cathodic chamber as high DO inhibits denitrification (Kelly and He, 2014). pH is also important because protons could be the limiting reagent during cathodic denitrification (Clauwaert et al., 2009). The pH could be increased to ensure sufficient protons for denitrification. A low C/N ratio is preferred because excess carbon could promote inhibit bioelectrochemical denitrification (Zhang and He, 2013). An analysis of the microbial community could reveal which microorganisms are present in the nitrification and cathodic chambers and indicate how environment conditions could be changed to promote nitrification and denitrification, respectively.

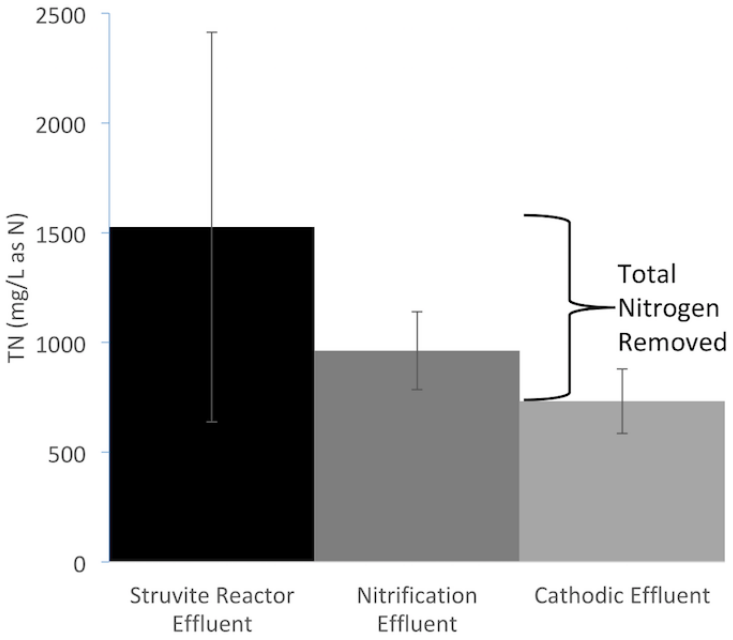

Figure 7. Concentration of Total Nitrogen (TN, mg/L as N) in struvite reactor effluent (stream 1 in Figure 6), fixed-film nitrification effluent (stream 2 in Figure 6), and cathodic effluent (stream 3 in Figure 6). Bar heights are arithmetic mean values of multiple point measurements taken over a 28-week period; error bars show plus or minus one standard deviation.

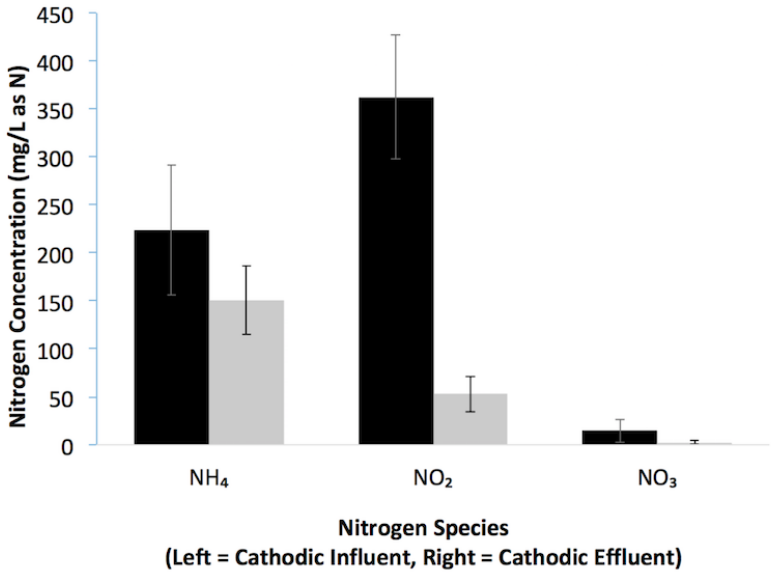

Figure 8. Nitrogen species in cathodic influent (stream 2 in Figure 6) and cathodic effluent (stream 3 in Figure 6). Bar heights are the arithmetic mean values of multiple point measurements taken over a 28-week period; error bars show plus or minus one standard deviation.

4.3.3 Performance of the Microbial Fuel Cell

In the anodic chamber, an average of 51% of the COD in the influent primary wastewater (270 ± 180 mg/L over the 28-week operation period) was removed. The COD removal was 45% during open circuit mode and 53% in closed circuit mode. The coulombic efficiency, which is defined as the percentage of electrons released by the electron donor that are recovered as current, was 18%. The enhancement of removal efficiency for the closed circuit condition and coulombic efficiency percentage provide evidence of electricity generation's role in COD removal. This indicated that organic matter was oxidized, releasing electrons which could be accepted by the anode and donated in the cathodic chamber. In theory, as shown in the denitrification equation below, an MFC with glucose as the electron donor in the anodic chamber and nitrate as the electron acceptor in the cathodic chamber has a standard electromotive force of approximately 1.2 V. If nitrite is the electron acceptor in the cathode instead of nitrate, the standard electromotive force is approximately 0.9 V.

- Denitrification:

Anode: $C_6H_{12}O_6 + 6\ H_2O \rightarrow 6\ CO_2 + 24\ H^+ + 24\ e^-$ $E^{0'}$ (V) = –0.43

Cathode: $2.4 * (2\ NO_3^- + 12\ H^+ + 10\ e^- \rightarrow N_2 + 6\ H_2O)$ $E^{0'}$ (V) = +0.75

Overall: $C_6H_{12}O_6 + 4.8\ NO_3^- + 4.8\ H^+ \rightarrow 6\ CO_2 + 2.4\ N_2 + 8.4\ H_2O$ $E^{0'}$ (V) = +1.18

- Denitritation:

Anode: $C_6H_{12}O_6 + 6\ H_2O \rightarrow 6\ CO_2 + 24\ H^+ + 24\ e^-$ $E^{0'}$ (V) = –0.43

Cathode: $4 * (2\ NO_2^- + 8\ H^+ + 6\ e^- \rightarrow N_2 + 4\ H_2O)$ $E^{0'}$ (V) = +0.44

Overall: $C_6H_{12}O_6 + 8\ NO_2^- + 8\ H^+ \rightarrow 6\ CO_2 + 4\ N_2 + 10\ H_2O$ $E^{0'}$ (V) = +0.87

In practice, the voltage range of MFCs with organic decomposition in the anodic chamber and oxygen as an electron acceptor in the cathodic chamber is commonly 300-700 mV due to losses associated with activation, bacterial metabolism, and mass transport (Logan et al. 2006). During the 28-day start-up period when glucose and sodium nitrate were used in the anodic and cathodic chambers, respectively, the average open circuit voltage was 83 mV. During the full implementation stage, the average open circuit voltage was 221 mV and the average closed-circuit voltage was 18 mV, which resulted in a calculated average current of 18 μA. Thus, the MFC generated 0.3 mW/m^2 (based on the surface area of the anode) or 1.1 mW/m^3 (based on the volume of liquid in each chamber of the MFC). This is less than the 8-12 mW/m^2 measured by Lee et al. (2013) using a similar setup to remove nitrogen and recover energy from diluted landfill leachate. The difference in power output is likely due to Lee's larger electrode surface area, different and diluted influent source, different bacterial community, and the frequent pulse input of leachate into the reactor. The two-chambered design, chosen for this proof-of-concept study to monitor and evaluate electron transfer, likely contributed to the low power output due to high internal resistance from membrane fouling and distance between electrodes. Future efforts will be made to improve the design to reduce internal resistance and increase power production by decreasing electrode spacing, reducing membrane fouling, and maintaining good contacts in the circuit (Logan et al. 2006). Power production may also depend on the concentrations of the electron donor or acceptor in the MFC.

Based on the observed COD reduction of 0.139 g/L in the anode, anodic chamber volume of 0.1 L, and the molar ratio of 4 moles of electrons per 32 g of COD, it was estimated that 0.0017 moles of electrons (estimated by multiplying the previous three values) were donated through organic decomposition in the anodic chamber over the hydraulic residence time of 19.4

days. Based on the average current of 18 μA (C/s), time duration of 19.4 days (1,676,160 s), and Faraday's constant of 96,500 C/mol, 0.0003 moles of electrons travelled through the wire during the same time period.

Interestingly, it appears that 0.0066 moles of electrons were accepted in the cathodic chamber through nitrite reduction (based on observed decrease in nitrite concentration from 360 mg/L NO_2^--N to 50 mg/L NO_2^--N), which was over three times the amount of electrons produced in the anodic chamber. This implies that a large amount of electrons are coming from somewhere other than the current. One possible explanation is that ammonium in the cathode may be donating electrons; it was observed that 0.094 g NH_4^+/L were also removed in the cathodic chamber, which would correspond to 0.0020 moles of electrons if the NH_4^+ was converted to N_2. However, the observed disappearance of NH_4^+ is still only able to explain about 40% of the "extra" denitritation observed, suggesting that some other unidentified electron-donating process was also occurring.

4.3.4 Evaluation of the Concept

Based on the nitrogen removal and energy generation performance, the proof-of-concept process merits further consideration in treating the liquid effluent of ESP. Results of this research demonstrated that fixed-film nitrification and an MFC can remove 52% of total nitrogen from the liquid effluent of ESP and generate 0.3 mW power per m^2 of anodic surface area. With additional refinement of the process, this may be an improvement over technologies such as ANAMMOX that remove up to 90% of ammonium but do not recover energy (van der Star et al. 2007). This may also be an improvement over implementing only struvite precipitation (without the MFC) because the proposed new technology achieves additional nitrogen removal and also recovers energy.

The treatment process as demonstrated is not energy-neutral, as the energy input (2.7 W) for the aerator is greater than the energy output of 3.3E-07 W from the MFC. However, this was a proof of concept study and at least three improvements can be tested to improve power output. First, aeration was not optimized in this study; 2.3 W of energy input for aeration was used only to produce nitrite, which requires less oxygen input than nitrate. Future MFC research can explore the possibility of using less energy input for aeration than the current study to see if similar amounts of nitrite and power are produced in the nitrification chamber. This can be done by using a controller to turn on aeration only when required. Second, other MFC designs can be employed that generate more power than the dual-chambered MFC design (Choudhury et al. 2017). One important factor that affects power output is electrode spacing; reducing electrode spacing reduces internal resistance and increases ionic diffusion rates (Janicek et al. 2014). The distance between electrodes in this dual-chambered MFC was 9 cm. Flat-plate and tubular reactors are often used in scaled-up studies because they can increase power output by limiting the spacing between electrodes to 1 cm or even 1 mm (Janicek et al. 2014). Third, improvements in materials and scalability can increase power output, which is probably best used to power locally situated equipment on-site. For example, tubular reactors can be operated continuously and be connected in series (Janicek et al. 2014) to increase power output.

4.4 Conclusions and Implications for Engineering Practice

Presently, the primary benefit of the MFC in the technology demonstrated here is not its ability to produce energy, but rather its ability to remove additional nitrogen; the sidestream nutrient removal prevents nutrients from returning to mainstream treatment, reducing operational costs. For example, by not returning nitrogen to the mainstream, a wastewater treatment plant could save the cost of treating ammonia ($0.61/kg NH_3) through aeration and avoid the cost of

methanol usage ($1.50/L), commonly used for denitrification (Drexler et al. 2014). If improvements are made to wastewater treatment processes, wastewater treatment can further transition to the resource recovery facility of the future by becoming a net-energy producer (Shoener et al. 2014) while also achieving the simultaneous benefits of nutrient recovery/removal and reduced costs associated with mainstream treatment. Besides the ESP applications to wastewater treatment plant sidestreams, urine, landfill leachate, and agricultural waste, this process can also be implemented to recover energy and reduce nutrient loading in natural (sediment, marine environments and lagoons) and industrial applications (Santoro et al. 2017).

CHAPTER 5: ASSESSMENT OF NUTRIENT FLUXES AND RECOVERY FOR A SWINE MANURE TREATMENT SYSTEM IN COSTA RICA

5.1 Introduction

The United Nations Sustainable Development Goal 6 contains specific targets that include providing access to adequate and safe sanitation for all, increasing recycling and safe water reuse, and improving water quality through pollution reduction. The lack of treatment of agricultural waste often leads to environmental problems such as eutrophication, greenhouse gas emissions, and health issues. A developing paradigm in wastewater treatment is that wastewater contains valuable resources, such as nutrients, energy, and water, that be economically recovered (e.g. fertilizer, energy, water) rather than simply treating contaminants (Verbyla et al., 2013; Water Environment Federation [WEF], 2014; Orner and Mihelcic, 2018). This resource recovery paradigm can contribute to the fulfillment of multiple Sustainable Development Goals (SDGs) related to food security and sustainable waste management (UN, 2018; Orner et al., 2017).

Agricultural waste in much of the world is typically not treated or, if treated, does not recover resources. For example, of 93,000 farms in Costa Rica (the location of this study), 79,000 have no treatment for their agricultural waste (Costa Rica Ministerio de Ambiente y Energia, 2015). A previous study in Costa Rica determined that agricultural waste directly contaminated rivers resulting in higher levels of bacteria and nutrients in the affected rivers (Shahady and Boniface, 2018). At farms in Costa Rica that do manage waste, commonly utilized technologies include small-scale digesters, sedimentation tanks, oxidation lagoons, and artificial

drains (Costa Rica Ministerio de Ambiente y Energia, 2015). If agricultural waste is land-applied without treatment, pathogens are not removed, and contaminants may runoff into nearby water bodies.

Opportunities exist to recover nutrients, energy, and water from agricultural waste. In particular, small-scale anaerobic digesters produce a biogas that can be used for producing electricity and/or heat, a cooking fuel, or a transportation fuel when processed and refined. The biosolids that accumulates in the digester can be harvested and used as a soil amendment. However, anaerobic digesters may not remove significant amounts of important pathogens (Manser et al., 2015, 2016). They also have high capital costs compared to lagoons and are not designed to remove or recover nutrients like nitrogen and phosphorus (Amini et al., 2017).

The mechanisms and efficiencies of removing nitrogen and phosphorus in such small-scale anaerobic digesters are not well understood, as the technology is primarily utilized for its recovery of energy from biogas. It has been estimated that 90% of organic nitrogen and phosphorus entering small-scale digesters can be converted into ammonia and phosphate, respectively (Song et al., 2011). The nutrient-rich liquid effluent leaving the anaerobic digester can then be land-applied (Zeng and Li, 2006). This is beneficial for plants, which prefer mineralized forms of nitrogen and phosphorus rather than the organic forms found in raw manure (Moser, 1998). Additionally, some nitrogen and phosphorus can be removed from the influent stream via transfer to the solid phase (sludge), but the rates and mechanisms of this transfer are not well understood. For example, Amini et al. (2017) measured no reduction of total nitrogen and moderate reduction (43%) of total phosphorus in a 30-L pilot scale digester, whereas a previous investigation of a tubular digester in Monteverde, Costa Rica, measured a high reduction in both total nitrogen (84%) and total phosphorus (92%) (Kinyua et al., 2016).

However, the solids were not studied and a mass balance was not constructed in either study to determine the mechanisms of nitrogen or phosphorus reduction. Therefore, small-scale anaerobic digesters offer benefits of biogas production and mineralization of nutrients, but the fate of N and P is, for the most part, not understood.

Besides small-scale anaerobic digestion, another technology that has been used to recover nutrients from a variety of waste streams (including agricultural waste) is the precipitation of struvite ($MgNH_4PO_4$) (Doyle and Parsons, 2002; Rahman et al., 2014). The recovery of ammonium and phosphate from struvite precipitation has been studied in several source streams including human urine (Etter et al., 2011; Ishii et al., 2015), landfill leachate (Gunay et al., 2008; Huang et al., 2014), industrial wastewater (Diwani et al., 2007; Matynia, 2013), anaerobic digester effluents (Celen, 2001; Munch & Barr, 2001), and swine wastewater (Suzuki et al., 2007; Liu et al., 2011; Amini et al., 2017). Targeted waste streams generally contain relatively high concentrations of ammonium and phosphate, in which case magnesium is the limiting reagent for precipitation of struvite; thus magnesium is added to obtain a 1:1:1 molar ratio of Mg:N:P. Because a pH of 8.5 or higher is recommended, base is also added for precipitation to occur (Stratful et al., 2001). The application to swine wastewater is especially promising as swine waste contains moderate levels of magnesium; therefore, less potentially costly magnesium addition is required to precipitate struvite (Dockhorn, 2009). Precipitating struvite from swine wastewater would also allow farmers to self-sufficiently produce a slow-release, nutrient-rich fertilizer that has similar properties to conventional synthetic fertilizer (Ahmed et al., 2006). Previous lab- and pilot-scale struvite studies using an influent of swine digester effluent found nitrogen recovery between 7.5% and 49% (Lind et al., 2000; Song et al., 2011;

Amini et al., 2017) and phosphorus recovery between 85% and 97% (Perera et al., 2007; Song et al., 2011; Amini et al., 2017).

However, existing struvite precipitation technologies using an influent of swine digester effluent require electricity and expensive equipment that may not be appropriate for small rural farms in low- and middle-income countries. For example, Ostara has developed a commercial struvite fertilizer design, but it is applicable only for large-scale wastewater treatment plants (Ostara, 2018). Previously, Etter et al. (2009) developed a low-cost struvite reactor design for human urine influent, but the design has yet to be tested for other influents such as swine digester effluent. It has not yet been determined if the nitrogen and phosphorus in swine digester effluent can be recovered through struvite precipitation using low-cost, locally available materials without the input of electricity.

Therefore, the overall goal of this chapter was to determine the ability of a treatment system for agricultural waste, comprising a tubular anaerobic digester that is integrated with a low-cost, locally produced struvite precipitation reactor, to contribute to the achievement of multiple Sustainable Development Goals during treatment. The three specific objectives were to 1) understand the efficiency and mechanisms of nutrient removal in two tubular digesters that receive agricultural waste by conducting a mass balance for nitrogen and phosphorus, 2) construct and assess a low-cost, locally produced struvite precipitation reactor that receives effluent from the two tubular digesters, and 3) understand the efficiency of nutrient removal in the low-cost, locally produced struvite precipitation reactor by conducting a mass balance for nitrogen and phosphorus. The treatment system could offer multiple benefits to a community: improved sanitation, removal of nutrients to prevent eutrophication, recovery of struvite as a potential fertilizer, and production of a final effluent stream that is suitable quality to be safely

used in aquaculture (Ostara, 2018). These are examples of how, more generally, quantifying nutrient recovery from agricultural waste and understanding recovery mechanisms can facilitate progress toward multiple sustainable development goals by improving sanitation, promoting sustainable management of wastes and natural resources, improving food security, and supporting local ecosystems.

5.2 Materials and Methods

5.2.1 Site Description

The study took place in the rural Costa Rican community of San Luis de Monteverde, a community of approximately 500 people who primarily work in agriculture and tourism. Farmers raise chickens, swine, and dairy cows and grow coffee, fruit, vegetables, and some medicinal plants. The University of Georgia-Costa Rica (UGA-CR), in addition to hosting researchers, tourists, and students studying abroad, maintains a working farm to provide food and energy for the campus and to test prototypes of developing technologies. The system for managing farm waste, shown in Figure 9, includes the treatment of feces from several swine and dairy cows using two digesters. Each morning at 6 AM (local time) a maintenance worker opens several valves to drain the swine waste by gravity into the two digesters. Then the maintenance worker uses a hose to sluice remaining large fecal matter from the pens to the digesters. Fecal matter from dairy cows is sluiced into digester #1.

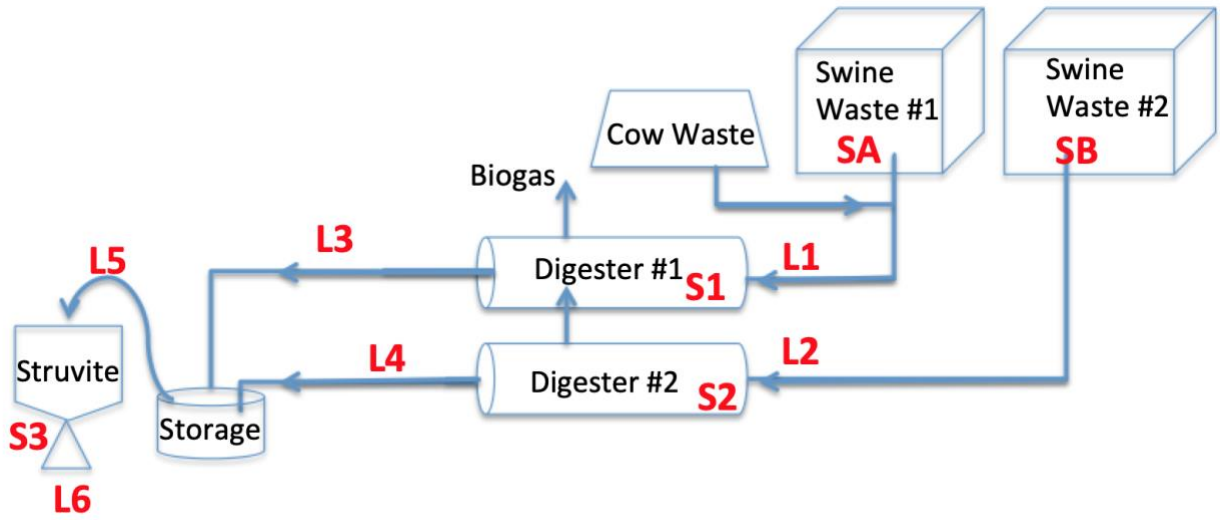

Figure 9. Process for resource recovery from agricultural waste using two tubular digesters and a struvite precipitation reactor. Abbreviations that begin with S and L are sampling locations for solids (S) and liquids (L), respectively.

5.2.2 Operation of Two Tubular Digesters

The average operating parameters of the two digesters are shown in Table 15. Digester #2 was built in 2016 when additional swine were purchased to handle the increased loading. The flow into digester #1 is greater than digester #2 because high volumes of water are used during and after milking of dairy cows. The effluent from both digesters flows by gravity to the storage containers.

Table 15. Tubular digesters' average operating parameters.

Parameter	Unit	Digester #1	Digester #2
Volume	L	12,000	12,000
Temperature of Digester Contents	°C	21.1	21.1
Flow	L/d	840	670
Hydraulic Residence Time	d	14.3	17.9
Organic Loading Rate	g VS/L-d	0.10	0.10

5.2.3 Construction and Operation of Struvite Reactor

For this study, I constructed a 200-L struvite precipitation reactor based on a previous design used for the precipitation of struvite from urine (Etter et al., 2011). Goals during construction were to use local appropriate materials and provide beneficial resources back to farmers. The cost of the reactor was approximately US$920, which included $660 for materials, $160 for labor, and $100 for transportation of materials. The reactor was operated in batch mode fourteen times between May and October 2018. The digester effluent is received in a storage container (covered with a cloth filter cover to reduce solids from entering) before being manually poured into the struvite precipitation reactor. Each batch typically contained 50 L of digester effluent, although up to 200 L of digester effluent could be poured into the struvite precipitation reactor for each batch.

Inducing the precipitation of struvite requires the addition of magnesium and a base. Therefore, 100 mL of bittern, a liquid byproduct from salt production (17g/L Mg, 11g/L K, 29g/L Na, 13g/L S), was added to provide sufficient magnesium. The bittern was obtained from a salt production facility located approximately 60 km away from UGA-CR. Because a common base like NaOH was not locally available and would have to be ordered from a specialty manufacturer, locally available CaO (quicklime) was initially used as the base. Five grams of CaO were added at a time into the reactor until a pH of at least 8.5 was reached. Because the introduction of calcium can produce calcium phosphate as a precipitate and thus minimize the production of the more desirable struvite, the base was switched in July 2018 to soda ash (Na_2CO_3), which is produced along with carbon dioxide and water when baking soda is heated. Soda ash is able to raise the pH in the reactor without adding calcium that might compete for phosphate.

Once the magnesium and base were added to the reactor, a handle attached to a stirring mechanism was rotated by hand at approximately 60 RPM for five minutes to stimulate mixing and precipitation of struvite. A filter bag made of manta, a cloth used in Costa Rica to collect coffee grounds before drinking coffee, was placed under the reactor to collect contents once the valve was opened. The cloth filtered struvite (and any other solids) from the liquid exiting the reactor through the effluent pipe. Effluent liquid emptied into a lagoon in which fish are raised. The filter bag was hung for drying near an air vent for one day, and then the struvite powder was removed from the bag with a brush and stored in a plastic container.

5.2.4 Sampling and Analysis

I collected liquid samples from the two tubular digesters and from the struvite precipitation reactor at the six locations shown in Figure 9. These samples were collected during 15 sampling campaigns approximately once every two weeks between February and October 2018. I analyzed liquid samples at UGA-CR for five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total solids (TS), volatile solids (VS), total suspended solids (TSS), and volatile suspended solids (VSS). I used Hach kits (Loveland, CO) to measure TN (TNT 827) and TP (TNT 845) using a PG InstrumentsT60 Visible Spectrophotometer (Leicestershire, United Kingdom) at the State Distance University (UNED) (San Jose, Costa Rica), and I measured COD (TNT 82206) with a Hach portable colorimeter (Loveland, CO). I measured BOD_5 (5210), TS, VS, TSS, and VSS using standard methods (APHA, 2012). Samples were filtered and analyzed to determine concentrations of anions and cations such as NO_3^-, PO_4^{3-}, Mg^{2+}, Ca^{2+}, and NH_4^+ at the UCR using Inductive Coupled Plasma Atomic Emission Spectroscopy (ICP-OES). I measured other water quality parameters such as pH, conductivity, temperature, dissolved oxygen, and NH_4^+ using a YSI multiprobe (Yellow Spring, OH).

Solid samples (swine manure, digester sludge, struvite) were collected and brought to the University of Costa Rica (UCR) (San Jose, Costa Rica) for analysis of percent solids, percent nitrogen, and percent phosphorus.

5.2.5 Estimation of Flow Rates and Concentrations

Flow rates and concentrations utilized in the mass balance were measured or estimated independently, as summarized in Table 16. The struvite reactor was operated in batch mode with a volume of 50 L. However, the mass balance for the struvite precipitation reactor utilized a volume of 1450 L to estimate the daily flux of nutrients (i.e. all digester effluent in one day was assumed to be treated in the struvite precipitation reactor).

Table 16. Descriptions, units, and estimations for symbols utilized in the mass balance equations.

Symbol	Description	Unit	How Estimated
Q_{SA}, Q_{SB}	Mass flow rate of swine waste	mg/d	Estimated by weighing fecal mass
Q_{S1}, Q_{S2}	Mass flow rate of digester sludge	mg/d	Estimated from mass balance
Q_{L1}, Q_{L2}	Volumetric flow rate of digester influent	L/d	Measured by filling 10 L bucket throughout day
Q_{L3}, Q_{L4}	Volumetric flow rate of digester effluent		Measured by filling 10 L bucket throughout day
M_{S3}	Mass of struvite	mg	Estimated from mass balance
V_{L5}, V_{L6}	Volume of liquid entering and leaving struvite reactor	L	Measured by filling 10 L bucket
TP_{SA}, TP_{SB}	Total Phosphorus concentration of swine waste	mg/g	Measured at UCR
TP_{S1}, TP_{S2}	Total Phosphorus concentration of digester sludge	mg/g	Measured at UCR
TP_{S3}	Total Phosphorus concentration of struvite	mg/g	Measured at UCR
TP_{L1}, TP_{L2}	Total Phosphorus concentration entering digester	mg/L	Measured with Hach TNT 845
TP_{L3}, TP_{L4}	Total Phosphorus concentration leaving digester	mg/L	Measured with Hach TNT 845
TP_{L5}, TP_{L6}	Total Phosphorus concentration entering and leaving struvite reactor	mg/L	Measured with Hach TNT 845
TN_{SA}, TN_{SB}	Total Nitrogen concentration of swine waste	mg/g	Measured at UCR
TN_{S1}, TN_{S2}	Total Nitrogen concentration of digester sludge	mg/g	Measured at UCR
TN_{S3}	Total Nitrogen concentration of struvite	mg/g	Measured at UCR
TN_{L1}, TN_{L2}	Total Nitrogen concentration entering digester	mg/L	Measured with Hach TNT 827
TN_{L3}, TN_{L4}	Total Nitrogen concentration leaving digester	mg/L	Measured with Hach TNT 827
TN_{L5}. TN_{L6}	Total Nitrogen concentration entering and leaving struvite reactor	mg/L	Measured with Hach TNT 827
$R_{digest,1}$, $R_{digest,2}$	**Rate of solids digestion**	**g/d**	**Estimated from mass balance**
TSS_{L1}, TSS_{L2}	TSS=Total Suspended Solids concentration entering digester	mg/L	Measured using Standard Methods
TSS_{L3}, TSS_{L4}	TSS=Total Suspended Solids concentration leaving digester	mg/L	Measured using Standard Methods

5.3 Results and Discussion

5.3.1 Phosphorus, Nitrogen, and Solids Mass Balances in Digesters

For each digester, the influent is a combination of the initial water and liquid waste (L1, L2 in Figure 9) that enters the digester by gravity when a valve is manually opened each morning

along with feces that enters the digester by washing (SA, SB). Some phosphorus, nitrogen, and solids from the two influents accumulate in the sludge of the digesters (S1, S2) while the remainder leaves in the digester effluent (L3, L4). Additionally, some solids are digested, converting organic phosphorus and nitrogen into phosphate and ammonium, respectively, as shown in Figure 10. Mass balance equations for digester #1, digester #2, and the struvite reactor are shown below in Table 17.

Multiple processes occur in the digester (shown in Figure 10), including digestion, biological assimilation, and precipitation (Metcalf & Eddy, 2014). These three terms are combined into the digestion term in the solids mass balance (Table 17, Equation 3). The solids (cells in the case of biological assimilation, or struvite or other precipitate in the case of chemical precipitation) are separated from the liquids through sedimentation. In the struvite reactor, chemical precipitation converts NH_4 and PO_4 into struvite, which is separated from the liquid via sedimentation and captured via filtration.

Table 17. Mass balance equations for digester #1, digester #2, and struvite reactor. Items in bold are the unknown variables.

Location	Equation #	Equation
Digester #1	1	$Q_{SA}\,TP_{SA} + Q_{L1}\,TP_{L1} = \mathbf{Q_{S1}}\,TP_{S1} + Q_{L3}\,TP_{L3}$
	2	$Q_{SA}\,TN_{SA} + Q_{L1}\,TN_{L1} = \mathbf{Q_{S1}}\,TN_{S1} + Q_{L3}\,TN_{L3}$
	3	$Q_{SA} + Q_{L1}\,TSS_{L1} = Q_{S1} + Q_{L3}\,TSS_{L3} + \mathbf{R_{digest,1}}$
Digester #2	4	$Q_{SB}\,TP_{SB} + Q_{L2}\,TP_{L1} = \mathbf{Q_{S2}}\,TP_{S2} + Q_{L4}\,TP_{L4}$
	5	$Q_{SB}\,TN_{SB} + Q_{L2}\,TN_{L1} = \mathbf{Q_{S2}}\,TN_{S2} + Q_{L4}\,TN_{L4}$
	6	$Q_{SB} + Q_{L2}\,TSS_{L2} = Q_{S2} + Q_{L4}\,TSS_{L4} + \mathbf{R_{digest,2}}$
Struvite Reactor	7	$V_{L5}\,TP_{L5} = \mathbf{M_{S3}}\,TP_{S3} + V_{L6}\,TP_{L6}$
	8	$V_{L5}\,TN_{L5} = \mathbf{M_{S3}}\,TN_{S3} + V_{L6}\,TN_{L6}$

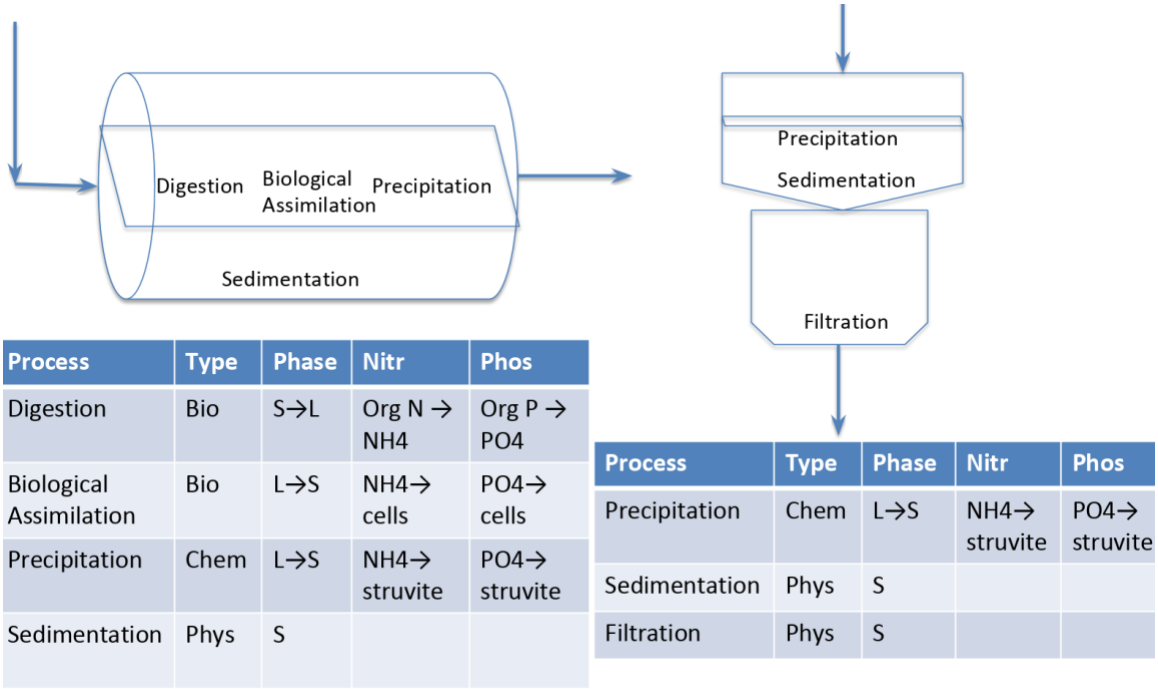

Figure 10. Biological, chemical, and physical processes occurring in tubular digesters and struvite precipitation reactor. Bio=Biological, Chem=Chemical, Phys=Physical, S=Solid, L=Liquid.

In equations 1-6, it is assumed that all N and P that enter the digester either exits in the effluent or is accumulated in the sludge. In other words, there are no loss mechanisms for N or P in the digesters (e.g. it is assumed that no N is lost to nitrification/denitrification). In contrast to N and P, the suspended solids may be lost through digestion. Equations 3 and 6 indicate that solids enter the digester in one of two ways (in the daily emptying of the swine waste or in the sluice that follows), and that the solids that enter can either settled to the sludge, exist as suspended solids in the effluent stream, or be broken down via digestion. In equations 1-6, the terms Q_{S1} (mg/d) and Q_{S2} (mg/d) represent the rate at which solids accumulate in the sludge layer of the respective digester. These settling rates cannot be directly measured and therefore must be estimated by solving equations 1, 2, 4, or 5. These variables are input into the solids balance, allowing the *digested* term (mg/d) to be determined in equations 3 and 6.

Average liquid concentrations for several water quality parameters that were input into the digester mass balance equations are found in Table 18.

Table 18. Tubular digesters' average influent and effluent characteristics between February and October 2018. The first number is the mean, and the number after "+/-" represents standard deviation. % Red is the percentage reduction from the influent mean to the effluent mean. Note: The liquid digester influent flow measurements do not represent all influent loadings as it does not include the solid feces terms (SA and SB).

			Digester #1			Digester #2			Average
Parameter	**Unit**	**n**	**Influent**	**Effluent**	**% Reduct - ion**	**Influent**	**Effluent**	**% Reduct - ion**	**% Reduct-ion**
TP	mg P/L	5	31.3 +/- 8.6	21.6 +/- 5.5	31%	47.6 +/- 16.5	52.8 +/- 9.8	-11%	10%
PO_4-P	mg PO_4-P/L	3	6.4 +/- 2.6	12.8 +/- 2.4	-100%	22.6 +/- 4.3	31.6 +/- 2.7	-40%	-70%
TN	mg N/L	3	142 +/- 36	120 +/- 40	15%	248 +/- 126	295 +/- 32	-19%	-2%
NH_4-N	mg NH_4^+-N/L	22	76 +/- 37	98 +/- 28	-29%	123 +/- 53	235 +/- 67	-91%	-60%
TS	g TS/L	11	2.2 +/- 1.6	0.8 +/- 0.4	61%	3.2 +/- 1.3	1.5 +/- 0.4	53%	57%
VS	g VS/L	11	1.4 +/- 1.3	0.4 +/- 0.2	70%	1.8 +/- 0.9	0.8 +/- 0.2	53%	61%
TSS	g TSS/L	10	1.1 +/- 0.8	0.2 +/- 0.2	83%	1.8 +/- 1.5	0.2 +/- 0.1	90%	87%
VSS	g VSS/L	10	0.7 +/- 0.4	0.2 +/- 0.1	74%	1.3 +/- 1.0	0.2 +/- 0.1	87%	81%
BOD_5	g/L	10	1.5 +/- 1.1	0.2 +/- 0.2	86%	1.3 +/- 0.6	0.3 +/- 0.2	79%	82%
COD	g/L	7	2.6 +/- 1.4	0.3 +/- 0.1	87%	2.8 +/- 1.3	0.6 +/- 0.1	80%	84%
pH		30	6.9 +/- 0.9	7.0 +/- 0.2		7.4 +/- 0.7	7.2 +/- 0.3		

Note: Negative percentages indicate increased values in the effluent due to, for example, digestion.

5.3.2 Phosphorus and Nitrogen Mass Balances in Struvite Precipitation Reactor

For the struvite reactor, there is one influent (L5) and two effluents as some phosphorus and nitrogen is precipitated into struvite (S3), while the remaining nutrients leave in the effluent (L6). Average liquid concentrations for several water quality parameters that were input into the mass balance equations for the struvite precipitation reactor are found in Table 19.

Table 19. Average influent and effluent characteristics in the struvite precipitation reactor between February and October 2018.

Parameter	Unit	Influent	Influent + Mg + Base	Effluent	% Reduction
TP	mg P/L	47.9	47.9	18.9	60%
PO_4-P	mg PO_4-P/L	24.3	24.3	5.2	79%
TN	mg N/L	205	205	185	10%
NH_4-N	mg NH_4^+-N/L	165	168	145	14%
pH		7.25		8.52	
TS	mg/L	1200		2700	

The TS, VS, TSS, and VSS decreased on average by 57%, 61%, 87%, and 81%, as shown in Table 16. The solids reduction values in the two tubular digesters in this study are similar to those of other tubular digesters in Costa Rica. In two studies by Lansing et al. (2008a, 2008b), TS, VS, and TSS removal percentages in tubular digesters were 67%, 83%, and 86%, respectively. A study by Kinyua et al. (2016) of digester #1 reported a TSS reduction of 86%. The solids results indicate that the tubular digesters are performing as expected and are effective in removing the majority of TSS from agricultural waste.

Table 20 provides information on percentage of solid constituents from field samples (digester and struvite reactor) and theoretical percentage (synthetic fertilizer, struvite, biomass). The N:P molar ratios of 9.4 (digester #1) and 9.9 (digester 2) indicate that any reduction of N and P in the sludge is likely due to biomass assimilation and sedimentation (high N:P) rather than struvite precipitation (1:1 N:P). The lower N% and P% in digester #1 is likely due to more inert material like clay and sand and less biomass. Because digester #1 receives influent from a concrete floor where cows are milked and swine are butchered and cooked over a wood fire, inert material like sand and ash is deposited and washed into digester #1.

Table 20. Percentage of constituents from field samples (digester, struvite reactor) and theoretical percentages (synthetic fertilizer, struvite, biomass).

Element	%Mg	%N	%P	%K	%Ca	%C	N:P molar ratio
Digester #1	0.3	2.8	0.7	0.3	1.5	26.8	9.4
Digester #2 (avg)	0.3	5.1	1.2	0.7	1.9	42.6	9.9
Struvite Reactor 1	1.7	0.9	3.1	1.7	26.2	12.4	0.6
Struvite Reactor 2	9.9	2.4	12.8	9.9	1.7	10.8	0.4
Synthetic Fertilizer		10	30	10			0.7
100% Struvite	9.8	5.7	12.7				1.0
Biomass ($C_5H_7O_2N$)		12.4				53.1	high

5.3.3 Fate of Phosphorus in the Two Tubular Digesters

Results from the mass balance for phosphorus in digester #1 and digester #2 are shown in Figure 11. The effluent phosphate was an average of 131% higher than the influent phosphate due to digestion and release of particulate P into soluble P; in digester 1, the effluent was 176% higher (4 to 11 g P/d), and in digester #2 the effluent was 86% higher (11 to 21 g P/d). Two other studies found an increase in phosphate concentration of 16% and 24% (Lansing et al., 2008a; Lansing et al., 2008b), about one quarter the increase found in this study. Because digester #2 most often receives influent from six larger swine and digester #1 receives influent from four smaller swine, the TP loading (g/d) in digester #2 (43 g P/d) is almost twice that of digester #1 (27 g P/d). Contrary to the increases in phosphate due to digestion, decreases in TP are caused by sedimentation. The effluent TP from digester #1 decreased 33% to 18 g P/d and from digester #2 decreased 18% to 36 g P/d. This value is close to the 36% decrease measured by Lansing et al. (2008b), and is less than the 91.6% percent TP reduction from a previous investigation of digester #1(Kinyua et al., 2016).

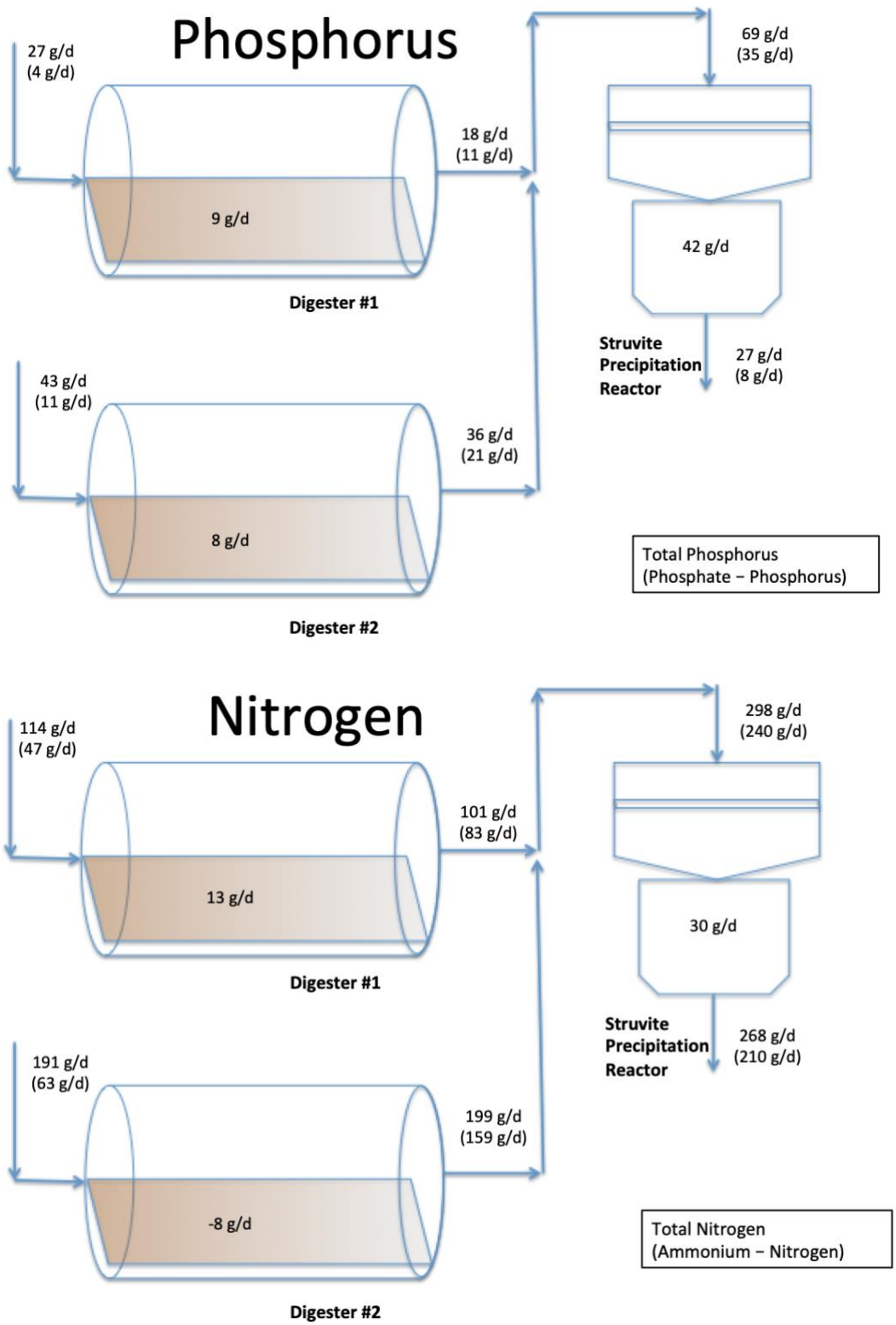

Figure 11. Fate of phosphorus (above) and nitrogen (below) in digester #1, digester #2, and struvite precipitation reactor.

5.3.4 Fate of Nitrogen in the Two Tubular Digesters

The ammonium increased from the digester influent to the effluent an average of 116%; digester #1 increased 78% (47 g NH_4-N/d to 83 g NH_4-N/d), whereas digester #2 had a much larger increase of 154% (63 g NH_4^+-N /d to 159 g NH_4^+-N /d). The two studies of small-scale tubular digesters in Costa Rica by Lansing reported 67% and 78% increases in ammonium, whereas a previous investigation of digester #1 found a 62% reduction in total ammonia nitrogen (TAN) (140 mg NH_4^+-N/L to 53 mg NH_4^+-N/L) and an 84% reduction in TN (300 mg N/L to 50 mg N/L) (Kinyua et al., 2016). Because digestion increases NH_4^+ concentrations by converting organic N into NH_4^+ and nitrification is not expected to convert NH_4^+ into NO_3^- due to lack of oxygen in the digester, an increase in NH_4^+ is more likely. Total nitrogen decreased by an average of 4%; in digester #1 TN dropped by 11% (114 g N/d to 101 g N/d), but increased in digester #2 by 4% (191 g N/d to 199 g N/d) likely due to resuspension of sludge or underestimation of the influent TN.

5.3.5 Fate of Phosphorus in the Struvite Reactor

During struvite precipitation, phosphate decreased by 79% from 35 g P to 8 g P. This is slightly under the range of 85-97% phosphorus recovery in previous studies that used an influent of swine digester effluent (Perera et al., 2007; Song et al., 2011; Amini et al., 2017). However, only 60% of TP was removed (69 to 27 g P). Of the 69 g of influent P, 35 g was in the form of phosphate. 42 g ended up in solid form while 27 g P remained in the effluent, 8 g in the form of phosphate (Figure 11).

An initial analysis of the solid pellets revealed a P% of 3.1 (Table 20). During this initial testing period, CaO was used to raise the pH, and coincidentally the Ca% in the solids collected in the cloth filter bag was 26.2%, meaning that a significant amount of calcium did not dissolve

into the digester effluent or precipitated with phosphate to form $CaPO_4$, which prevents the formation of struvite and subsequent removal of nitrogen from the digester effluent. This was apparent in the low Mg% (1.7%). Therefore, starting in July 2018, a base of soda ash was used instead of CaO to prevent $CaPO_4$ formation in further tests. Results from the second solid struvite analysis revealed a much lower Ca% (1.7%) and a much higher Mg% (9.9%). The P% also rose from 3.5% in the first test to 12.8% in the second test. The molar ratio of Mg:P of 1:1 in the second test also indicates struvite formation. However, the second test had a C% of 10.7%, indicating that some organics were mixed together with the struvite.

5.3.6 Fate of Nitrogen in the Struvite Reactor

During struvite precipitation, the ammonium decreased by 12% from 240 g N to 210 g N. This is slightly less than the hypothesis of 15% N recovery. However, the removal percentage varied based on what type of cloth filter bag was used. Initially, a cloth filter bag with small pores was used. However, the bag didn't allow liquid to pass easily, resulting in the tearing of the filter bag. A second filter bag was used that had larger pores, but the larger gaps allowing solids to pass through in addition to the liquid. Therefore, the first cloth filter bag material was used (manta), but the liquid was delivered at a much slower flow rate. The 12% ammonium removal is within the range of 7.5% to 49% ammonium removal found in previous struvite precipitation studies using an influent of swine digester effluent (Lind et al., 2000; Song et al., 2011; Amini et al., 2017). TN decreased by 10% from 298 g to 268 g. A first analysis of the solid struvite pellets revealed an N% of 1.1 (Table 12). The low percentage was likely due to the CaO that didn't dissolve (26.2% Ca). The second analysis of the solid struvite pellets, conducted when using soda ash as a base, revealed an increased N% of 2.4%.

Synthetic agricultural fertilizers, which are often used by Costa Rican farmers, are produced by mining nutrients typically needed for encourage healthy soils and plant growth. The synthetic fertilizer typically contains nutrient percentages of nitrogen, phosphorus, and potassium (NPK) in the ratio of 10/30/10. Pure struvite fertilizer has an NPK of 6/13/0. Results from the first solid struvite test revealed an NPK of 1/3/2. Given the low NPK values and the fact that precipitation of $CaPO_4$ (instead of $MgNH_4PO_4$) would not recover nitrogen, use of CaO as a base was not further pursued. Results from the second solid struvite test using soda ash as a base revealed an NPK of 2/13/10, which has less nitrogen than pure struvite and less nitrogen and phosphorus than the synthetic fertilizer. It is hypothesized that ammonia volatilization was occurring as the pH of the liquid struvite effluent was above 8.5. The 10% recovered potassium indicates that K-struvite ($KMgPO_4$-$6H_2O$) may also have been recovered, but this typically occurs only once nitrogen has been depleted (Jagtap and Boyer, 2018). Approximately 5 g of solid were recovered for every 50 L of liquid entering the struvite reactor.

5.4 Conclusions

The overall goal of this study was to determine the ability of a small-scale tubular digester and a low-cost, locally produced struvite precipitation reactor used to treat agricultural waste to facilitate the achievement of multiple sustainable development goals. Results indicate that an average of 25% of P and 4% of N was removed in the digesters through sedimentation. The digesters also promoted digestion, as seen by the increase in PO_4-P and NH_4-N concentrations by 131% and 116%, respectively. Additionally, 79% of PO_4-P and 12% of NH_4-N were removed from the liquid effluent of the two digesters in a struvite precipitation reactor using low-cost, locally available materials. Overall, if struvite precipitation was implemented

full-scale, TN would decrease by 70% and TP would decrease by 13% from the digester influent to the struvite effluent.

The recovered solid in the struvite reactor appears to be struvite as the Mg:P ratio is 1:1; however, the recovered solid's N/P/K ratio of 2.5/12.9/9.9 is less than the current synthetic fertilizer's N/P/K ratio of 10/30/10. However, struvite is less soluble than synethetic fertilizer, meaning that nutrients are disseminated to the plants over a longer period of time, reducing the amount of fertilizer needed. One recommendation to promote struvite collection and reduce tearing is to slowly drain the struvite reactor effluent through the cloth filter bag.

One conclusion from the study is the need to properly manage the effluent products of the tubular digester. Just as releasing the biogas causes increased amounts of damaging methane to the environment, releasing the liquid digester effluent directly to water bodies results in increased amounts of potentially damaging NH_4 and PO_4 to the aquatic environment. Because these mineralized forms of nitrogen and phosphorus are more easily utilized than their organic forms by plants, harmful algae blooms could be more likely if the effluent is directly released to water bodies. Therefore, there is a need to take up the NH_4 and PO_4 prior to entering water bodies, and struvite precipitation is a viable technology to uptake these mineralized nutrients.

One promising result of this study was the development of a low-cost, locally produced struvite precipitation reactor that uses locally available materials for influents. The reactor was built on site using local labor and local materials for approximately $920. Costs could be further decreased through bulk purchases. A magnesium source, which is typically a barrier to struvite production due to its high cost, was found as a waste product called bittern from a nearby salt production facility. Likewise, a base was produced from soda ash by heating locally bought baking soda. These local products allow for the production of struvite, a fertilizer that can

replace synthetic fertilizer for rural farmers, while also reducing nutrient loading into local water bodies, protecting the environment and the people who make a living from ecotourism.

Stuvite is currently being produced from anaerobic digester effluent with no detectable pathogens (<3 MPN/g) by Ostara at several centralized wastewater treatment plants in the United States and Europe (Ostara, 2018a). Ostara's process includes rinsing the struvite with non-potable water, dewatering on a screen that produces a 80% solids product, and then drying at 93 °C to reduce moisture content below 0.5%. Most pathogens are inactivated during dewatering, and the remainder are inactivated during drying (Ostara, 2018b).

Several studies have investigated the fate of pathogens from struvite produced from urine. Schurmann et al. (2012) found no bacteria in urine struvite after washing with a saturated struvite solution and drying at 30 °C. Bischel et al. (2016) found that bacteria are most effectively inactivated by heating the struvite to at least 50 °C under humid conditions, followed by a drying step. A human virus surrogate (Phage ΦX174) was inactivated in urine struvite due to reduced moisture content during drying at mild temperatures (Decrey et al., 2011). Helminths, which are prevalent in tropical and sub-tropical climates, can accumulate in the cloth filter bag. A helminth surrogate (Ascaris suum) was inactivated in urine struvite due to reduced moisture content (36% relative humidity) and elevated drying temperatures (above 35 °C); however, temperatures above 55 °C may cause loss of nitrogen due to ammonia volatilization (Decrey et al., 2011).

Blackwater has more pathogens than urine. Gell et al. (2011) produced struvite from both urine and blackwater, and found no bacteria in the struvite produced from either influent, perhaps due to high salinity levels (Muller et al., 2006). Pathogens nor indicators were tested in this study of struvite precipitation from agricultural waste. Based on the studies above, it is expected that

pathogens in struvite produced from agricultural waste can be reduced by heating the struvite at 50 °C under humid conditions followed by dewatering and drying. Care should be taken to avoid pathogen transmission throughout the sanitation service chain. For example, personal protective equipment should be used by operators filtering the struvite bags to avoid contact with digester effluent, which may contain pathogens (Bischel et al., 2016).

The struvite precipitation process could be improved in three ways. First, the struvite reactor could be sited at a lower elevation so that the storage container would be at a higher elevation than the top of the struvite reactor. This would eliminate the need to manually lift the struvite influent from the storage container to the top of the struvite reactor and would enable continuous filling of the reactor. Second, urine, which is currently mixed with feces in septic tanks, could be source-separated and collected to increase phosphorus concentrations entering the digester and produce more struvite. Third, I recommend having multiple filter bags available to enable continuous processing and lids for the struvite storage container and struvite precipitation reactor to prevent rain or leaves from entering.

Overall, a tubular digester and a low-cost, locally produced struvite precipitation reactor can facilitate progress toward multiple Sustainable Development Goals (SDGs) related to sustainable management of wastes and natural resources and improve food security by improving yields from community agriculture practices that also support local ecosystems.

CHAPTER 6: OVERALL CONCLUSIONS AND RECOMMENDATIONS

6.1 Evaluating Wastewater Nutrient Management Technologies using a House of Quality Planning Matrix

The removal and recovery of nutrients from wastewater can contribute to achieving multiple SDGs such as SDG 6 (Clean Water and Sanitation), SDG 2 (Zero Hunger) and SDG 12 (Responsible Consumption) and multiple NAE Grand Challenges such as managing the nitrogen cycle and providing clean water). Because of population growth and urbanization, nutrient management will occur at an increasing number of centralized WWTPs. Nutrients can be managed for different types of influents (e.g. domestic and agricultural). Given that nutrients from domestic wastewater are more concentrated at the building- and community-scales, opportunities exist to recover nutrients using a variety of technologies at a variety of scales within a sewershed (see Figure 2). However, decision-making on managing nutrients can be difficult due to a wide variety of priorities to consider. A House of Quality planning matrix that considered priorities at three scales in a sewershed produced rankings that generally align with current wastewater treatment practice. At the city scale, top-ranked technologies are those commonly employed (e.g. A^2O, oxidation ditch) that use the dissolved organic carbon present in the wastewater to drive denitrification, and conventional treatment is ranked highest at the building scale because of its easy maintenance, small footprint, and inoffensive aesthetics. However, future trends will likely affect the technologies, weightings, and scores and therefore change the ranking of the technologies. For example, continued research and development of

wastewater treatment technologies, especially at the building and community scales, may result in higher scores across all ten technical characteristics for up-and-coming technologies, thus disrupting the status quo of a flush toilet connected to a septic tank or sewer system. This trend may be amplified by the implementation of test beds, which can provide opportunities to improve the technical characteristics of developing technologies while minimizing risk for municipalities.

The House of Quality planning tool was utilized to analyze nutrient management technologies at three scales across a sewershed, using the sewershed of Hillsborough County Northwest Facility near Tampa, FL as a case study. Results indicate that building-scale source separation and community-scale nutrient management could reduce nitrogen loading to the mainstream treatment train of the centralized wastewater treatment plant by over 50%, but sidestream treatment has minimal impact in nitrogen reduction. Conversely, sidestream treatment technologies such as ion exchange are the most effective in reducing phosphorus loading to the mainstream due to high quantities of phosphorus recycling back to the head of the plant; building- and community-scale technologies are moderately effective in reducing phosphorus loading to the treatment plant. These results can inform decision-makers what context-specific nutrient management technologies to consider at a variety of scales, and illustrate that sidestream technologies are most effective in reducing phosphorus loading while building- and community-scale technologies are most effective in reducing nitrogen loading to the centralized treatment plant.

6.2 Recovering Energy and Managing Nitrogen from Struvite Precipitation Effluent via Microbial Fuel Cells

Two technologies that together manage nutrients and recover energy in sidestreams at centralized WWTPs are struvite precipitation and microbial fuel cells. Because the liquid effluent from engineered struvite precipitation still contains high quantities of nitrogen, a fixed-film nitrification reactor and a two-chambered MFC can further reduce nitrogen and recover energy. Presently, the primary benefit of the MFC in the technology demonstrated here is not its ability to produce energy, but rather its ability to remove additional nitrogen; the sidestream nutrient removal prevents nutrients from returning to mainstream treatment, reducing operational costs. If improvements are made to wastewater treatment processes, wastewater treatment can further transition to the resource recovery facility of the future by becoming a net-energy producer (Shoener et al. 2014) while also achieving the simultaneous benefits of nutrient recovery/removal and reduced costs associated with mainstream treatment.

6.3 Recovering Nutrients, Energy, and Water from Agricultural Waste via Small-Scale Tubular Digesters and a Struvite Precipitation Reactor

In agricultural settings, especially in developing countries, energy and nutrients can potentially be recovered by an agricultural waste treatment system comprising a small-scale tubular anaerobic digester integrated with a low-cost, locally produced struvite precipitation reactor. In the two digesters, liquid-phase concentrations of PO_4^{3-}-P and NH_4^+-N increased by averages of 131% and 116%, respectively, due to release from the manure during anaerobic digestion. Despite this increase in liquid-phase concentrations, an average of 25% of total phosphorus and 4% of total nitrogen was removed from the influent swine manure through sedimentation in the digesters. During struvite precipitation, an average of 79% of PO_4^{3-}-P and

12% of NH_4^+-N was removed from the waste stream and produced a solid with percentages of Mg, N, P of 9.9%, 2.4%, and 12.8%, respectively, indicating that struvite was likely formed. I facilitated construction of a struvite precipitation reactor that was built on site using local labor and local materials for approximately $920. Local products such as bittern (magnesium source) and soda ash (base) allowed for the production of struvite, a fertilizer that can replace synthetic fertilizer for rural farmers. The treatment system offers multiple benefits to the local community: improved sanitation, removal of nutrients to prevent eutrophication, recovery of struvite as a potential fertilizer, and production of a final effluent stream that is suitable quality to be used in aquaculture. These are examples of how, more generally, quantifying nutrient recovery from agricultural waste and understanding recovery mechanisms can facilitate progress toward multiple sustainable development goals by improving sanitation, promoting sustainable management of wastes and natural resources, improving food security, and supporting local ecosystems.

6.4 Moving Forward

Removing and recovering nutrients from a variety of influents at a variety of scales can contribute to the achievement of multiple sustainable development goals. Decision-makers need effective technologies to manage nutrients and decision-making tools to assist in the process of choosing those technologies at a variety of scales. For example, development of easy-to-use models to understand the fate of nitrogen and phosphorus at a sewershed level is needed, especially with the depletion of phosphorus reserves. Technologies can be implemented at a variety of scales, and municipalities have to balance multiple priorities while minimizing risk. These priorities can be advanced by continuing technology development at test beds. The implementation of technologies where nutrients are most concentrated (i.e. building-scale) may

require changes in behavior or the involvement of small-scale businesses. Integrating multiple disciplines such as anthropology, business, microbiology, and public health is needed to reduce health risk and manage nutrients through implementing infrastructure, mobilizing small businesses, and changing behavior in a culturally sensitive way. As the trends of population growth and resource depletion continue, wastewater will increasingly be seen as a fountain of resources such as nutrients, energy, and water that can contribute to the achievement of multiple SDGs.

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APPENDIX B. ADDITIONAL TABLES FOR "A CASE STUDY FOR ANALYZING NUTRIENT MANAGEMENT TECHNOLOGIES AT THREE SCALES WITHIN A SEWERSHED"

Table B1. House of Quality for building scale.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Weighting		1.2	1.2	1	1	0.5	0.9	1.1	1.1	1.1	0.9	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SDF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	Composting Toilet		-	--	0	+	0	+	++	+	+	-	31.8
	Conventional Wastewater Treatment		++	+	0	+	--	++	--	--	--	+	29.7
	Direct Urine Application as Fertilizer*		--	--	--	+	+	0	++	+	0	-	27.1
	Treatment wetlands		-	--	--	+	-	+	+	-	-	++	27.0
	Urine ANAMMOX*		--	--	--	0	0	-	+	++	++	-	26.9
	Aerobic MBR		--	0	--	0	++	+	0	-	-	0	25.3
	Septic systems		+	-	-	0	--	++	--	--	--	+	24.1
	Anaerobic MBR		-	0	--	0	+	0	0	--	--	0	22.9
	Anaerobic Digestion*		-	--	--	0	0	+	-			--	22.4
	Struvite + Absorption with Zeolites in Urine*		--	--	--	-	+	-	0	+	+	--	22.2
	Struvite Precipitation in Urine*		--	--	--	-	++	-	+	--	++	--	21.6
	Nitrification and Distillation of Urine*		--	--	--	0	0	-	+	++	--	--	21.6
	NH3 Stripping to H2SO4 in Urine*		--	--	--	-	+	-	+	++	--	--	21.1
	Anion Exchange in Urine*		--	--	--	--	0	-	0	--	++	--	18.5

An * denotes the requirement of source separation of urine

Table B2. House of Quality for community scale.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Weighting		1.2	1.2	1	1	0.5	0.9	1.1	1.1	1.1	0.9	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SDF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	Constructed Wetlands		++	--	++	++	-	++	0	0	0	++	37.1
	Biological Nutrient Removal (BNR)		--	0	-	-	+	++	0	++	+	0	31.2
	Facultative Lagoons		++	--	+	+	--	++	0	0	-	-	30.8
	Algal Photobioreactors		-	-	-	-	+	0	+	+	++	0	30.5
	Sequencing Batch Reactors - BNR mode		--	0	-	-	+	+	0	+	++	0	30.3
	Recirculating Sand Filter		0	-	0	+	--	++	0	0	-	+	30.4
	Rotating Biological Contactor		-	0	-	-	+	++	0	0	0	0	29.1
	Intermittent Sand Filter		+	-	0	+	--	++	0	-	-	0	29.6
	Communal Septic Systems		++	-	+	++	--	++	-	--	--	0	29.5
	Membrane Bioreactors - BNR mode		--	0	--	--	++	+	0	+	++	0	28.8
	Membrane Bioreactors - BNR mode		--	0	--	--	++	+	0	+	++	0	28.8
	Algal Membrane Bioreactors		-	-	--	--	++	-	+	+	++	0	28.1
	Aerated Lagoons		+	--	0	+	--	++	0	0	--	0	28.4
	Activated Sludge Systems		-	0	0	0	0	++	0	-	-	0	28.4
	Upflow Anaerobic Sludge Blanket		-	0	0	++	+	+	0	--	--	0	27.8
	Anaerobic Lagoons		++	--	+	++	--	++	-	--	--	-	27.4
	Anaerobic Membrane Bioreactors		-	0	--	+	+	0	0	--	--	0	23.9

Table B3. House of Quality for city scale sidestream.

			O&M	SF	CC	OC	EP	TM	EI	PN	PP	A	
	Weighting		1.2	1.2	1	1	0.5	0.9	1.1	1.1	1.1	0.9	
Technical Characteristics	Ease of Operation & Maintenance	O&M											
	Size/Density/Footprint	SDF											
	Capital Cost	CC		✓									
	Operational Cost	OC	✓										
	End Products	EP											
	Technical Maturity	TM	✓		x	x	✓						
	Environmental Impact	EI		✓									
	Performance N	PN	✓				✓	✓	x				
	Performance P	PP	✓				✓	✓	x				
	Aesthetics	A		✓	✓		✓						
Technologies	Struvite Precipitation*		0	+	-	-	++	++	+	-	++	0	35.3
	No treatment		-	++	0	++	-	++	++	--	--	+	33.2
	Ion Exchange		-	+	--	-	++	0	++	+	0	0	31.3
	Forward osmosis		0	0	--	-	-	-	++	++	+	0	31.1
	Conventional nitrification-denitrification*		0	0	-	-	-	++	+	++	--	0	30.4
	Ammonia stripping*		0	0	-	-	+	+	+	+	--	0	30.3
	Microbial fuel cell		0	0	--	0	0	0	+	++	--	0	29.1
	Shortcut nitrogen removal*		-	0	-	-	-	++	+	+	--	0	28.1
	Breakpoint chlorination		-	-	-	-	--	+	+	+	--	0	25.5

An * denotes the requirement of a previous step of anaerobic digestion

APPENDIX C. TRAINING MANUAL FOR STRUVITE PRECIPITATION REACTOR IN MONTEVERDE, COSTA RICA

Manual de Construcción y Operación del Reactor de Estruvita

Construction and Operation Manual For Struvite Precipitation Reactor

por Kevin Orner
by Kevin Orner

Noviembre 2018
November 2018

Como hacer fertilizante

#1 Preparar Reactor Influente

a. Coloque la cuberta del filtro de tela sobre el contendedor de almacenamiento.
b. Conecte los codos de PVC a los tubos de salda de PVC del efluente del digestor para comenzar a fluir a través de la cubierta del filtro de tela hacia el contenedor de almacenamiento.
c. Permitir que el contenedor de almacenamiento se llene a un mínimo de 50 litros.

#2 Se Llene el Reactor

a. Cierre la válvula de salida.
b. Cargar 50 litros de efluente del digestor del contenedor de almacenamiento. Use una cubeta de 10 litros para subir el efluente del digestor por la escalera del reactor y hacia la parte superior del reactor.

#3 Añadir Magnesio (bittern)

a. Agregue 100 ml de bittern en un recipiente cerrado.
b. Vaciar 100 ml de bittern en la parte superior del reactor.

#4 Añadir Base (Ceniza de Soda)

a. Añadir una media botella de ceniza de soda en el reactor.

#5 Precipitar Estruvita

a. Use el mango para agitar inmediatamente después de la adición de magnesio para evitar su sedimentación.
b. Revuelva durante 5 minutos.

#6 Vacie el Reactor

a. Coloque la bolsa de filtro de tela en el reactor de estruvita.
b. Abra la válvula de salida para permitir que el efluente fluya hacia la bolsa de filtro de tela. No abra la válvula completamente para mantener la tasa de salida a una tasa aceptable.
3. Espere hasta que el reactor de estruvita se vacíe completamente.

#7 Cosecha la Estruvita

a. Cierre la válvula de salida y separe la bolsa del filtro.
b. Cuelgue la bolsa del filtro para que se seque.
c. Después de que la bolsa esté completamente seca, el polvo de estruvita se puede quitar fácilmente.
d. Limpie la bolsa del filtro, que se reutilizará para el siguiente lote.

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1 Introduccion / *Introduction*

Los residuos agrícolas no tratados pueden perjudicar la salud ambiental, causando eutrofización, emisiones de gases de efecto invernadero y problemas de saneamiento. De 93,000 granjas en Costa Rica, 79,000 no tienen tratamiento para sus residuos agrícolas. Un estudio previo realizado por Shahady y Boniface en Costa Rica determinó que los desechos agrícolas contaminaban directamente los ríos y producían niveles más altos de bacterias y nutrientes.

El tratamiento de los desechos agrícolas no solo puede reducir estos problemas, sino que también presenta una oportunidad para recuperar recursos beneficiosos como los nutrientes, la energía y el agua. En particular, los digestores anaeróbicos a pequeña escala producen un biogás que se puede usar para generar electricidad y / o calor, como combustible para cocinar o como combustible para el transporte cuando se procesa y se refina. El lodo que se acumula en el digestor también se puede utilizar como enmienda del suelo.

Los digestores anaeróbicos no están diseñados para eliminar o recuperar nutrientes como el nitrógeno y el fósforo. Sin embargo, aproximadamente el 90% del nitrógeno orgánico y el fósforo que entran en los digestores a pequeña escala se pueden convertir en amoníaco y fosfato, respectivamente. El efluente líquido rico en nutrientes que sale del digestor anaeróbico puede aplicarse en tierra. Esto es beneficioso para las plantas, que prefieren las formas mineralizadas de nitrógeno y fósforo en lugar de las formas orgánicas que se encuentran en el estiércol crudo. Además, parte del nitrógeno y el fósforo se pueden eliminar de la corriente de entrada mediante la transferencia a la fase sólida (lodo).

Además de la digestión anaeróbica a pequeña escala, otra tecnología que se ha utilizado para recuperar nutrientes de una variedad de flujos de desechos (incluidos los desechos agrícolas) es la precipitación de estruvita ($MgNH_4PO_4$). Las corrientes de desechos dirigidas generalmente contienen concentraciones relativamente altas de amonio y fosfato, en cuyo caso el magnesio es el reactivo limitante para la precipitación de estruvita; así se agrega magnesio para obtener una relación molar 1: 1: 1 de Mg: N: P. Debido a que se recomienda un pH de 8.5 o superior, también se agrega una base para que ocurra la precipitación. La aplicación a aguas residuales porcinas es especialmente prometedora, ya que los residuos porcinos contienen niveles moderados de magnesio; por lo tanto, se requiere una adición de magnesio menos costosa para precipitar la estruvita. La precipitación de estruvita de las aguas residuales de los cerdos también permitiría a los agricultores producir de manera autosuficiente un fertilizante de liberación lenta y rico en nutrientes que tiene propiedades similares al fertilizante sintético convencional.

Por lo tanto, se desarrolló un sistema de tratamiento para residuos agrícolas, que comprende dos digestores anaeróbicos tubulares a pequeña escala que se integran con un reactor de precipitación de estruvita de producción local de bajo costo. En los dos digestores, las concentraciones en fase líquida de PO_4^{3-}-P y NH_4^+ -N aumentaron en promedios de 131% y 116%, respectivamente, debido a la liberación del estiércol durante la digestión anaeróbica. A pesar de este aumento en las concentraciones de la fase líquida, se eliminó un promedio del 25% del fósforo total y el 4% del nitrógeno total del estiércol de cerdo influyente a través de la sedimentación en los digestores. El efluente de los digestores se trató en un reactor de precipitación de estruvita que se construyó y operó utilizando materiales disponibles localmente y de bajo costo, tales como agua de remolacha (como fuente de magnesio) y ceniza de sosa (para elevar el pH). El reactor de precipitación de estruvita eliminó un promedio de 79% de PO_4^{3-}-P y 12% de NH_4^+ -N de la corriente de desechos y produjo un sólido con porcentajes de Mg, N, P de

9.9%, 2.4% y 12.8%, respectivamente. , lo que indica que probablemente se formó estruvita. Se produjeron aproximadamente 5 g de estruvita por cada 50 l de influyente.

Los fertilizantes sintéticos para la agricultura, que a menudo son utilizados por los agricultores costarricenses, son producidos por los nutrientes de la minería que generalmente se necesitan para fomentar la salud de los suelos y el crecimiento de las plantas. El fertilizante sintético típicamente contiene porcentajes de nutrientes de nitrógeno, fósforo y potasio (NPK) en la proporción 10/30/10. El fertilizante de estruvita puro tiene un NPK de 6/13/0. Los resultados de la prueba de estruvita sólida revelaron un NPK del 13/2/10, que tiene menos nitrógeno que la estruvita pura y menos nitrógeno y fósforo que el fertilizante sintético. Por lo tanto, 2-3 veces la cantidad de fertilizante de estruvita puede ser necesaria para lograr la misma masa de NPK que los fertilizantes sintéticos.

En general, este sistema de tratamiento ofrece múltiples beneficios a la comunidad local: saneamiento mejorado, eliminación de nutrientes para prevenir la eutrofización, recuperación de estruvita como fertilizante potencial y producción de un flujo final de efluentes que es de calidad adecuada para ser utilizado en la acuicultura. Estos son ejemplos de cómo la recuperación de nutrientes de los desechos agrícolas puede facilitar el progreso hacia múltiples objetivos de desarrollo sostenible al mejorar el saneamiento, promover el manejo sostenible de desechos y recursos naturales, mejorar la seguridad alimentaria y apoyar los ecosistemas locales.

Table: Percentage of Constituents from Field Samples (Digester, Struvite Reactor) and Theoretical Percentages (Synthetic Fertilizer, Struvite, Biomass)

Element	%Mg	%N	%P	%K	%Ca	%C	N:P molar ratio
Digester #1	0.3	2.8	0.7	0.3	1.5	26.8	9.4
Digester #2 (avg)	0.3	5.1	1.2	0.7	1.9	42.6	9.9
100% Struvite	9.8	5.7	12.7				1.0
Struvite Reactor	9.9	2.4	12.8	9.9	1.7	10.8	0.4
Synthetic Fertilizer		10	30	10			0.7

Untreated agricultural waste can impair environmental health, causing eutrophication, greenhouse gas emissions, and sanitation issues. Of 93,000 farms in Costa Rica, 79,000 have no treatment for their agricultural waste. A previous study by Shahady and Boniface in Costa Rica determined that agricultural waste directly contaminated rivers resulting in higher levels of bacteria and nutrients.

Treating agricultural waste not only can reduce these problems but also presents an opportunity to recover beneficial resources such as nutrients, energy, and water. In particular, small-scale anaerobic digesters produce a biogas that can be used for electricity and/or heat, a cooking fuel, or a transportation fuel when processed and refined. The sludge that accumulates in the digester can be also used as a soil amendment.

Anaerobic digesters are not designed to remove or recover nutrients like nitrogen and phosphorus. However, approximately 90% of organic nitrogen and phosphorus entering small-scale digesters can be converted into ammonia and phosphate, respectively. The nutrient-rich liquid effluent leaving the anaerobic digester can then be land-applied. This is beneficial for plants, which prefer mineralized forms of nitrogen and phosphorus rather than the organic forms

found in raw manure. Additionally, some nitrogen and phosphorus can be removed from the influent stream via transfer to the solid phase (sludge).

Besides small-scale anaerobic digestion, another technology that has been used to recover nutrients from a variety of waste streams (including agricultural waste) is the precipitation of struvite ($MgNH_4PO_4$). Targeted waste streams generally contain relatively high concentrations of ammonium and phosphate, in which case magnesium is the limiting reagent for precipitation of struvite; thus magnesium is added to obtain a 1:1:1 molar ratio of Mg:N:P. Because a pH of 8.5 or higher is recommended, base is also added for precipitation to occur. The application to swine wastewater is especially promising as swine waste contains moderate levels of magnesium; therefore, less potentially costly magnesium addition is required to precipitate struvite. Precipitating struvite from swine wastewater would also allow farmers to self-sufficiently produce a slow-release, nutrient-rich fertilizer that has similar properties to conventional synthetic fertilizer.

Therefore, a treatment system for agricultural waste was developed, comprising two small-scale tubular anaerobic digesters that are integrated with a low-cost, locally produced struvite precipitation reactor. In the two digesters, liquid-phase concentrations of PO_4^{3-}-P and NH_4^+-N increased by averages of 131% and 116%, respectively, due to release from the manure during anaerobic digestion. Despite this increase in liquid-phase concentrations, an average of 25% of total phosphorus and 4% of total nitrogen was removed from the influent swine manure through sedimentation in the digesters. Effluent from the digesters was treated in a struvite precipitation reactor that was constructed and operated using low-cost, locally available materials such as bittern (as a magnesium source) and soda ash (to raise the pH). The struvite precipitation reactor removed an average of 79% of PO_4^{3-}-P and 12% of NH_4^+-N from the waste stream and produced a solid with percentages of Mg, N, P of 9.9%, 2.4%, and 12.8%, respectively, indicating that struvite was likely formed. Approximately 5 g of struvite were produced for every 50 L of influent.

Synthetic agricultural fertilizers, which are often used by Costa Rican farmers, are produced by mining nutrients typically needed for encourage healthy soils and plant growth. The synthetic fertilizer typically contains nutrient percentages of nitrogen, phosphorus, and potassium (NPK) in the ratio of 10/30/10. Pure struvite fertilizer has an NPK of 6/13/0. Results from the solid struvite test revealed an NPK of 2/13/10, which has less nitrogen than pure struvite and less nitrogen and phosphorus than the synthetic fertilizer. Therefore, 2-3 times the amount of struvite fertilizer may be needed to achieve the same mass of NPK as synthetic fertilizers.

Overall, this treatment system offers multiple benefits to the local community: improved sanitation, removal of nutrients to prevent eutrophication, recovery of struvite as a potential fertilizer, and production of a final effluent stream that is suitable quality to be used in aquaculture. These are examples of how recovering nutrients from agricultural waste can facilitate progress toward multiple sustainable development goals by improving sanitation, promoting sustainable management of wastes and natural resources, improving food security, and supporting local ecosystem

2 Construcción /*Construction*

2.1 Materiales / *Materials*

Item	descripcion	grueso	area de superficie	pagina
hoja de metal galvanizado	reactor parte arriba	1.5 mm	615x1895 (D = 600 mm)	04-200-02
	reactor afilada	1.5 mm	156*, r694, arca 1885 mm	04-200-03
	reactor salida	1.5 mm	120 x h100 (d 38 mm)	04-200-01
	reactor tapa	1.5 mm	r305-r15 (x2)	04-200-02
	cuchilla de agitacion 1	3 mm	100x400	04-300-01
	cuchilla de agitacion 2	3 mm	100x400	04-300-01
	cuchilla de agitacion 3	3 mm	100x200	04-300-01
	strip para revolver	5 mm	25x280	04-300-01
	barras de fijacion para mecanismo de agitacion	5 mm	25x (50+300+300+50) (x2)	04-300-02
	strip para plataforma	5 mm	25x(75+25) (x2)	04-500-01
	parte de la bolsa de filtro	5 mm	70 x 70 (curvar esquinas r10)	04-600-01
tubo de metal redondo	manija	D = 3/8"	H = 110 mm	04-300-01
	mecanismo de agitacion	D = 3/8"	H = 150 mm	04-300-01
	revolvador	D = 12 mm	H = 1300 mm	04-300-01
	soporte del reactor - arriba	D = 1"	L = 1727 mm curvada (d 550)	04-400-01
	soporte del reactor - abajo	D = 1"	L = 3142 mm curvada (d 1000)	04-400-01
	soporte del reactor - piernas	D = 1"	L = 2007 mm (x6)	04-400-01
	soporte del reactor - escaleras	D = 1"	L <550 mm (x6) (x2)	04-400-01
	soporte de la bolsa de filtro	D = 8 mm	L = (35 mm + 35 mm) (x2)	04-600-01
tubo de metal cuadrado	plataforma - vertical	25x25x2 (1" x 1")	L = 1525 (x2)	04-500-01
	plataforma - horizonal	25x25x2 (1" x 1")	L = 470 (x3)	04-500-01
	plataforma - diagonal	25x25x2 (1" x 1")	L = 940	04-500-01
	plataforma - arriba	25x25x2 (1" x 1")	L = 800 (x2)	04-500-01

	plataforma -pedazos	25x25x2 (1" x 1")	L = 50 (x2)	04-500-01
plywood		15 mm	520 x 570	04-500-01
tubo de PPR				
		D = 50, t = 5.6 mm	L = 175 mm	04-600-01
		D = 50, t = 5.6 mm	L = 100 mm	04-600-01
valvula de bola PPR		D = 50		04-600-01
enchufe macho PPR		D = 50 x 3/2" thread		04-600-01
tuerca para encajar rosca en el enchufe macho		3/2" thread		04-600-01
bisagras				04-100-01
arandela		D = 3/8"		04-300-01
silicon glue				04-100-01
cartilla de acero				04-300-01, 04-300-02
pintura de alta calidad				04-300-01, 04-300-02
cubo de colleccion				04-100-01
filtro de nylon - lados		r1000-r500, 72*	L = 600 mm (arcas de 1251 y 500)	04-700-01
filtro de nylon - abajo		D = 400 mm		04-700-01
filtro de nylon - coleccion				-
manos para bolsa de filtro		D = 40 mm		04-600-01
lazo de tela			20 mm x 90 mm	04-700-01
cubo			v = 5 gal	-
tanque de coleccion			v = 200 L	-

2.2 Diseño/*Design*

Por favor, vea los apéndices para los documentos de diseño.
Please see the appendices for the design documents.

Figura 1: Alineación general
Figure 1: *General Arrangement*

Figura 2. Arriba a la Izquierda) Mecanismo de agitación. **Arriba a la Derecha)** Soporte del reactor, **Fondo)** Reactor

Figure 2. Top Left) *Stirring Mechanism*, ***Top Right)*** *Reactor Stand*, ***Bottom)*** *Reactor*

3 Operación / *Operation*

3.1 Materiales / *Materials*

Los dos ingredientes necesarios para operar la precipitación de estruvita son 1) bicarbonato de sodio y 2) agua asada.

The two ingredients needed to operate the struvite precipitation are 1) baking soda and 2) bittern.

Figura 3. Izquierda) bicarbonato de sodio, **Derecha) agua asada**

Figure 3. Left) *baking soda,* ***Right)*** *bittern*

3.1.1 Bicarbonato de sodio / *Baking Soda*

El bicarbonato de sodio debe colocarse en una lata de pastel y calentarse durante al menos una hora a 100 ° C. Cuando se calienta como tal, el bicarbonato de sodio se convierte en ceniza de sosa, dióxido de carbono y agua. La ceniza de sosa tiene un pH más alto y se usa como ingrediente para el reactor de precipitación de estruvita.

*The baking soda should be placed in a pie tin and heated for at least one hour at 100*C. When heated as such, the baking soda is converted into soda ash, carbon dioxide, and water. The soda ash has a higher pH, and is used as an ingredient for the struvite precipitation reactor.*

$$2NaHCO_3(s) \rightarrow Na_2CO_3(s) + CO_2(g) + H_2O(g)$$

3.1.2 Agua Asada / *Bittern*

Bittern se obtuvo de una instalación de producción de sal en la ciudad de Colorado, cerca del Golfo de Nicoya. Durante la producción de sal, uno de los subproductos es Bittern, un líquido rico en magnesio. Esto fue traído de vuelta al campus UGA-CR.

Bittern was obtained from a salt production facility in the town of Colorado near the Gulf of Nicoya. During salt production, one of the byproducts is bittern, a liquid rich in magnesium. This was brought back to the UGA-CR campus.

3.2 Procedimiento / *Procedure*

3.2.1 Preparar Reactor Influente / *Prepare Reactor Influent*

1. Coloque la cuberta del filtro de tela sobre el contendedor de almacenamiento. *Place cloth filter cover over storage container*

2. Conecte los codos de PVC a los tubos de salda de PVC del efluente del digestor para comenzar a fluir a través de la cubierta del filtro de tela hacia el contenedor de almacenamiento. *Connect PVC elbows to Digester Effluent PVC outflow pipes to begin flow through cloth filter cover into storage container*

3. Permitir que el contenedor de almacenamiento se llene a un mínimo de 50 litros. *Allow storage container to fill at a minimum of 50 liters.*

3.2.2 Se llene el Reactor / *Fill Reactor*

1. Cierre la válvula de salida.
Close outlet valve.

2. Cargar 50 litros de efluente del digestor del contenedor de almacenamiento. Use una cubeta de 10 litros para subir el efluente del digestor por la escalera del reactor y hacia la parte superior del reactor.
Load 50 liters of digester effluent from storage container. Use a 10-liter bucket to bring digester effluent up the reactor ladder and into top of reactor.

3.2.3 Añadir Magnesio (bittern) / *Add Magnesium (Bittern)*

1. Agregue 100 ml de bittern en un recipiente cerrado.
Add 100 mL of bittern into closed container.

2. Vaciar 100 ml de bittern en la parte superior del reactor.
Dump 100 mL bittern into top of reactor.

3.2.4 Añadir base (ceniza de soda) / *Add Base (Soda Ash)*

1. Añadir 5 g de ceniza de soda en el reactor.
Add 5g of soda ash into reactor.

2. Medir el pH del líquido en el reactor usando multiprobe.
Measure pH of liquid in reactor using multiprobe.

3. Repita hasta que el pH esté entre 8.5 y 9.
Repeat until pH is between 8.5 and 9.

3.2.5 Precipitar Estruvita / *Precipitate Struvite*

1. Use el mango para agitar inmediatamente después de la adición de magnesio para evitar su sedimentación.
Use the handle to stir starting immediately after magnesium addition to avoid its sedimentation.

2. Revuelva durante 5 minutos.
Stir for 5 minutes.

3.2.6 Vacie el Reactor / *Empty Reactor*

1. Coloque la bolsa de filtro de tela en el reactor de estruvita.
Place cloth filter bag on struvite reactor.

2. Abra la válvula de salida para permitir que el efluente fluya hacia la bolsa de filtro de tela. No abra la válvula completamente para mantener la tasa de salida a una tasa aceptable.
Open the outlet valve to let the effluent flow into the cloth filter bag. Do not open the valve fully to keep the outflow rate at an acceptable rate.

3. Espere hasta que el reactor de estruvita se vacíe completamente.
Wait until the struvite reactor is fully emptied.

3.2.7 Cosecha la Estruvita / *Harvest Struvite*

1. Cierre la válvula de salida y separe la bolsa del filtro.
Close the outlet valve and detach the filter bag.

2. Cuelgue la bolsa del filtro para que se seque. *Hang up the filter bag for drying.*

3. Después de que la bolsa esté completamente seca, el polvo de estruvita se puede quitar fácilmente.
After the bag is completely dry, the struvite powder can be easily removed.

4. Limpie la bolsa del filtro, que se reutilizará para el siguiente lote.
Clean the filter bag, which will be reused for the next batch.

ABOUT THE AUTHOR

Kevin Orner is a Ph.D. Candidate in Environmental Engineering at the University of South Florida, where he studies nutrient and energy recovery from wastewater. After obtaining a B.S. in Civil and Environmental Engineering with a certificate in Technical Communication from the University of Wisconsin-Madison in 2008, he served for two years as a Peace Corps Volunteer in Panama. In December 2011, he completed his M.S. in Civil and Environmental Engineering at the University of South Florida. He is an E.I.T. with engineering consulting experience. In 2018, Kevin conducted research under a Fulbright Research Grant in Costa Rica to investigate nutrient and energy recovery from pig and cow manure using anaerobic digestion and struvite precipitation.