QUANTIFYING SUSTAINABILITY IN WASTEWATER TREATMENT: EXAMPLES IN ALGACULTURE

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ABSTRACT

In the wastewater industry the concept of sustainability addresses how complex systems, aimed at mitigating or preventing pollution, involve tradeoffs in life cycle impacts. Advanced treatment systems designed for high quality effluent often come at the cost of increased chemical and energy use, and alternatives which minimize the impacts in all categories are desirable. Algaculture is a promising technology for sustainable wastewater treatment, but quantifying the impacts of these systems is prudent before they are implemented on a large scale. This work enhances the growing body of research on the topic, contributing assessments of algaculture wastewater treatment systems using model-based life cycle assessment (LCA), laboratory investigations, and data-based LCA.

The integration of algaculture into conventional activated sludge systems was investigated. Process modeling was used as the basis of a comparative LCA to determine environmental impacts. Integrating algaculture prior to activated sludge proved to be beneficial for all impact categories considered; however, this scenario would also require primary sedimentation and impacts of that unit process should be considered for implementation of such a system.

Membrane photo-bioreactors are proposed for use in algae-based wastewater to achieve nutrient removal with a relatively small footprint compared to other algaculture systems, but membrane fouling is problematic in these systems. Laboratory-scale bioreactors and membrane filtration procedures were used to determine the impact of nutrient limitation and culture density on fouling. Nitrogen limitation was found to exacerbate membrane fouling, and it is proposed that accumulation of carbon-rich intracellular metabolites and subsequent diffusion from cells was the mechanism observed.

Lagoon systems, once a common method of wastewater treatment which has fallen out of favor partially due to their unreliable ammonia control, can be retrofitted with rotating algal biofilm reactors (RABR) to improve treatment. This hybrid system was compared to an activated sludge systems in terms of operational, construction stage, and avoided life cycle impacts. Results show that the lagoon with an RABR system reduced eutrophication impacts more than the activated sludge system. Additionally, the resulting increase in global warming potential and cumulative energy demand for the RABR system was smaller than that of the activated sludge system.

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Aerobic tank	AER
Algal nutrient removal	ANR
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Algogenic organic matter	AOM
Anaerobic tank	ANA
Anoxic tank	ANX
Biological nutrient removal	BNR
Biological oxygen demand	BOD
Cellular organic matter	COM
Chemical oxygen demand	COD
Clean water flux	CWF
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Conventional nutrient removal	CNR
Cumulative energy demand	CED
Dissolved organic carbon	DOC
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Distilled, deionized	DDI
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Extracellular organic matter	EOM
Extant electricity for aeration	AEE
First, second, third quartile	Q_1, Q_2, Q_3
Global warming potential	GWP
Hydraulic retention time	HRT
Interquartile range	IQR
Land use	LU
Life cycle assessment	LCA
Membrane photobioreactor	MPBR
Microfiltration	MF
Million gallons per day	MGD
Molecular weight	MW
Monte Carlo analysis	MCA
Net environmental benefit	NEB
Optical density	OD
Photobioreactor	PBR
Primary algal nutrient removal	PANR
Primary energy demand	PED
Publicly owned treatment works	POTW
Recovery flux	RF
Rotating algal biofilm reactor	RABR
Rotating biological contactor	RBC

List of Abbreviations (continued)	
Sedimentation basin	SED
Sidestream algal nutrient removal	SANR
Sludge volume index	SVI
Solids retention time	SRT
Supplementary information	SI
Tertiary algal nutrient removal	TANR
Total nitrogen	TN
Total organic carbon	TOC
Total phosphorus	TP
Total dissolved solids	TDS
Total solids	TS
Total suspended solids	TSS
Triacylglycerides	TAG
Ultrafiltration	UF
Utah Department of Environmental Quality	UDEQ
Utah State University	USU
Wastewater treatment	WWT
Wastewater treatment plant	WWTP

CHAPTER ONE

INTRODUCTION

Sustainability is gaining increasing attention in popular culture and is often presented as a synonym for "environmentally friendly." However, research in the area of sustainability is more complicated than simply mitigating or preventing pollution; it involves working to understand how complex systems interact to impact the environment, society, and the economy. This concept as it relates to the wastewater industry is described by the tradeoffs of treatment; although there are economic costs associated with treating wastewater, the benefits to human and environmental health outweigh these costs. Additionally, advanced treatment aimed at improving local water quality is often accomplished through increased chemical and energy use at plants, which in turn negatively impact the global climate. Research efforts toward a more sustainable industry aim to mitigate these tradeoffs through technologies that can achieve energy and resource recovery to improve both local and global environmental health while minimizing economic impacts. Algaculture is a promising technology for sustainable wastewater systems that may help plants achieve improved treatment, energy efficiency, and resource recovery.

Background and Motivation

Algaculture in wastewater treatment

Historically, algae have played a large role in simple, low-tech wastewater treatment systems such as lagoons and oxidation ditches, providing oxygen for heterotrophic bacteria and partial nutrient removal by assimilation. However, recent interest in combined algaculture/wastewater treatment systems has arisen as a result of studies showing that algal biofuels may not be energy-positive in many cases when industrial fertilizers are used for biomass production (Beal et al., 2012; Clarens et al., 2010; Sander and Murthy, 2010). In addition to the benefits to a potential algal biofuel industry, there are also ways that algaculture may make wastewater treatment more sustainable. Low cost, less energy intensive nutrient removal is one appeal of algaculture in wastewater (Pittman et al., 2011). The ability of algae to remove metals from wastewater (Mehta and Gaur, 2005) may also lessen ecotoxicity impacts associated with metals in effluent or land applied biosolids (Foley et al., 2010; Godin et al., 2012), but the end-use of algal biomass and resulting impacts must also be considered. For algaculture to be incorporated into wastewater treatment facilities with the intent of aiding in treatment and producing an algal biomass product, it is important to understand how these systems can be integrated with existing infrastructure and how they will perform, particularly in terms of effluent quality.

There are several proposed strategies for the implementation of algaculture for wastewater treatment. Studies have looked at stand-alone algaculture plants (e.g. high-rate algal ponds (Park et al., 2011)) used to treat wastewater, potentially reducing the energy requirements for treatment. Although these systems have proven effective for treatment, they are limited to new plant construction which is rare in the United States. Additionally, uncertainties in performance of full-scale algaculture systems add risk to implementation of these processes. Instead, upgrades to existing plants are more common, which may include creating hybrid plants using both algaculture and more conventional treatment techniques.

Sustainability Assessment in Wastewater Treatment

Life cycle assessment (LCA) is a useful tool for evaluating wastewater treatment systems and the tradeoffs between local water quality and larger-scale environmental impacts observed as a result of advanced treatment (Corominas et al., 2013; Godin et al., 2012) and can complement the usual cost-based systems analysis used in conventional engineering practices. Impacts such as energy use and aquatic emissions can be directly compared to show cross-media effects of wastewater treatment (Foley et al., 2007) and may highlight areas in which algaculture improves sustainability of these systems. LCA may be used to assess the potential end-uses of algal biomass, through system expansion due to the wide range of LCAs published about algal biofuels and bioproducts (Benemann et al., 2012; Sander and Murthy, 2010), which may prove to enhance the appeal of such systems. This method should be approached with caution (Weidema, 2000); as system boundaries are expanded, the specificity of the study can be lost.

Some previous LCA studies of algal bioproducts have performed system expansion to include wastewater treatment offsets, but these are typically limited to a generalized energy savings based on nitrogen utilization (Clarens et al., 2010; Resurreccion et al., 2012). These studies do not include effects to other treatment operations and generally assume sufficient nutrient concentrations, which may not always be the case. Despite these limitations, existing algae-centric LCAs that assume nutrients are supplied via wastewater suggest a potential reduction in eutrophication impacts from algae systems, a mutual goal of a wastewater treatment plant. In addition, algaculture has the potential to diminish energy requirements at the plant, depending on the integration configuration. This, in turn, also diminishes global warming and other impacts associated with electricity production.

Research Objectives

The main goal of this work was to quantitatively assess how utilization of algaculture can affect the life cycle impacts of wastewater treatment. The specific objectives were (1) to apply theoretical models of algal and wastewater processes in a life cycle assessment framework to compare the performance of various hybrid algaculture/activated sludge treatment systems, (2) to perform laboratory experiments to understand how algal biomass separation processes (identified as a crucial process during work on the first objective) might impact an algaculture system's life cycle performance, and (3) to use lab- and pilot-scale data in conjunction with the life cycle modeling techniques used in the first objective to compare the performance of hybrid algaculture/lagoon and conventional activated sludge treatment systems.

The first objective was achieved using a combination of a stoichiometry-based algae model and activated sludge modeling. Integration of algal growth was modeled within various flows at a small, nitrifying activated sludge treatment plant. Eutrophication, global warming, primary energy demand, ecotoxicity, and land use impacts were evaluated. This work is presented in chapter 2, and served as background and motivation for the other objectives addressed.

Algal biomass removal, or harvesting, processes were modeled as ideal systems in chapter 2 to focus on the influence of algae biomass growth on activated sludge processes, therefore the impacts of these harvesting processes were not fully addressed; however it was concluded in this work that these processes will be crucial in ensuring that implementation of algaculture wastewater treatment is environmentally beneficial. Additionally, harvesting is often one of the largest hurdles to overcome in creating a feasible algal biomass production system. Therefore it is necessary to address harvesting in algaculture systems analysis, which is the aim of the second objective (chapter 3). Microfiltration was the separation technique studied in chapter 3, chosen because of its reliability in terms of effluent quality and its ability to reduce the footprint of algaculture systems, which were identified in chapter 2 as important considerations for eutrophication and land use impacts. This work aimed to elucidate the relationship between algal culture conditions and microfiltration performance in terms of membrane fouling, and was completed using laboratory-scale bioreactors and membrane filtration procedures. By understanding how the changes that occur in the biochemical environment of wastewater as a result of algal nutrient removal influence downstream separation processes, this work complements chapter 2 and gives a better understanding of the challenges and life cycle impacts related to the integration of algaculture in wastewater treatment.

Upon completion of chapters 2 and 3, it was found that the difficulties to overcome in using suspended growth algaculture at activated sludge treatment plants are significant; therefore an alternative concept for integrated algculture/wastewater treatment was assessed during completion of the third objective in chapter 4. This new approach addressed the shortcomings of high land use and biomass harvesting obstacles identified in the previous chapters. Chapter 4 focused on utilizing algal biofilm reactors (which allow for simpler harvesting techniques than suspended growth algal reactors) as an upgrade scenario to existing lagoon wastewater treatment plants (which are more land intensive than activated sludge systems). While many of the same techniques used in chapter 4 were similar to those described in chapter 2, this work also differs from the original study in the modeling of both the algal system (by using lab- and pilot- scale data rather than theoretical, stoichiometric relationships to model the system) and the activated sludge system (by addressing the uncertainty of the system as it relates to influent quality). This approach is an improvement to the theoretical models as it is more informative of the reality of both algal and conventional treatment systems in the future.

Each of the chapters that address an objective (chapters 2 through 4) includes literature review related to that work, a summary of the methods used, a description of the results found, and discussion of the relevance to the wastewater industry. Chapter 5 summarizes the conclusions from each chapter and provides recommendations for future research and practical applications. Supplementary information for chapters 2 through 4 is also included in the appendices.

CHAPTER TWO

INTEGRATING ALGACULTURE INTO SMALL WASTEWATER TREATMENT PLANTS: PROCESS FLOW OPTIONS AND LIFE CYCLE IMPACTS

Abstract

Algaculture has the potential to be a sustainable option for nutrient removal at wastewater treatment plants. The purpose of this study was to compare the environmental impacts of three likely algaculture integration strategies to a conventional nutrient removal strategy. Process modeling was used to determine life cycle inventory data and a comparative life cycle assessment was used to determine environmental impacts. Treatment scenarios included a base case treatment plant without nutrient removal, a plant with conventional nutrient removal, and three other cases with algal unit processes placed at the head of the plant, in a side stream, and at the end of the plant, respectively. Impact categories included eutrophication, global warming, ecotoxicity, and primary energy demand. Integrating algaculture prior to activated sludge proved to be most beneficial of the scenarios considered for all impact categories; however, this scenario would also require primary sedimentation and impacts of that unit process should be considered for implementation of such a system. This study has been published in *Environmental Science: Processes and Impacts* (Steele et al., 2014).

Introduction

Research and practice in the wastewater treatment field has shifted from strictly environmental protection to energy and resource recovery. Biogas and land-applied biosolids from anaerobic digestion are the most common methods of energy and resource recovery, but application of anaerobic digestion is often limited to large facilities. For small systems there remains a need to identify technologies that can accomplish net energy savings and resource recovery. Decreasing nutrient loadings in receiving waters has also become an important goal of wastewater treatment, especially "leading edge" methods employing biological nutrient removal (BNR). While improving local water quality by limiting nutrient emissions, BNR requires high energy demands for aeration, which increases greenhouse gas emissions (Foley et al., 2010, 2007). Alternate processes with low energy requirements are desirable.

Algaculture is one promising means of capturing and utilizing wastewater resources such as water, nitrogen, phosphorus, and carbon dioxide. Wastewater-fed algaculture is receiving a great deal of attention (Olguín, 2012). Much of the recent literature is devoted to creating biofuels, since it has been emphasized that fertilizer consumption in stand-alone algal biofuel production facilities is a serious impediment (Laurent Lardon et al., 2009). The use of wastewater to provide nutrients is one potential path forward toward making algal biofuels sustainable (Clarens et al., 2010; Pittman et al., 2011), thus the focus has been on whether the wastewater can support algal production. In that scenario the algae simply use the wastewater stream with no consideration of feedback to the wastewater treatment plant (WWTP). It is interesting to consider a different question: whether the use of algaculture can in some way enhance wastewater treatment. Clearly the algae could remove nutrients to improve effluent water quality, but could they also change the behavior of other unit processes to realize some synergistic benefits? This would be a true *integration* of algaculture and wastewater treatment.

One angle for accomplishing WWTP/algaculture integration is to mix algae with bacterial processes in the same tank for combined organic carbon and nutrient removal (de Godos et al., 2010; Medina and Neis, 2007; Muñoz et al., 2004), sometimes called "activated algae" (Mcgriff and McKinney, 1972). This follows from decades-old work showing that photosynthetic algae can potentially provide enough oxygen for heterotrophic bacteria to perform their function (Oswald et al., 1957). That approach has some promise, but may require an entirely new WWTP—or a complete overhaul—to create the algal/bacterial reactors, with very different hydraulic and solids retention times than existing plants.

Another angle for integrating algae with wastewater treatment is to keep the algaculture as a separate unit process, but place it at some location in the treatment train (or perhaps a side stream). This would be advantageous if an existing plant were being upgraded, as opposed to greenfield construction. Now that WWTPs are ubiquitous (at least in the developed world) most current construction projects are devoted to upgrades. Having an algal process that can be integrated during such an upgrade is the most likely way in which algaculture will be feasible for small systems in the near future.

There are three main locations in a conventional WWTP where an algaculture unit process could be added. The most commonly discussed location is at the end of the plant, where treated effluent is fed to algae as a polishing step to remove nutrients while growing algae for biofuel. This can be called "tertiary algaculture." Another likely location for algaculture implementation is at the head of the plant, treating raw or settled wastewater. In this "primary algal treatment" approach the algae not only utilize wastewater nutrients, but can also use organic carbon to increase algal biomass production (given an appropriate species). The remaining likely location for an algaculture unit process can be called "side-stream algaculture." This refers to the water produced in solids thickening operations, which can impart up to 30% of the plant's total nitrogen load, depending on the biosolids digestion operation. References for studies using each of the three wastewater types can be found in Table 2.1.

Table 2.1: References used to model nitrogen and phosphorus removal efficiencies for various wastewater streams and algal culture types

WW Type	Culture Type	Removal reported in terms of…	Reference
Treated	Mixed, Biofilm	NO_3^-, TP	(Boelee et al., 2011)
	Mixed, Biofilm	TN, TP	(Christenson and Sims, 2012)
	Muriellopsis sp.	NH_3, TP	(Gómez et al., 2013)
	Chlorella vulgaris	NH_3, NO_3^-, PO_4^{3-}	(He et al., 2013)
	Chlorella sorokiniana	NH_3	(Lim et al., 2013)
	Scenedesmus sp.	NH_3, PO_4^{3-}	(Zhang et al., 2008)*
	Mixed, *Scenedesmus sp.*	NH_3, TP	(Di Termini et al., 2011)
	Mixed, Algae/Sludge	NH_3, PO_4^{3-}	(Velasquez-Orta, 2013)
	Chlorella sp.	TN, TP	(Wang et al., 2013)
	Neochloris oleoabundans	NO_3^- , TN, TP	(Wang and Lan, 2011)
Untreated	*Euglena sp.*	NH_3, TN, TP, PO_4^{3-}	(Mahapatra et al., 2013)
	Mixed, *Chlorella vulgaris*/Sludge	TN	(Medina and Neis, 2007)
	Scenedesmus sp.	NH_3, TP	(Zhang et al., 2008)*
	Chlorella sp.	NH_3, TP	(Li et al., 2011)*
	Scenedesmus obliquus, Biofilm	NH_3, PO_4^{3-}	(Ruiz-Marin et al., 2010)*
	Mixed, *Chlorella sp.*	NH3, NO3-, and TP	(de-Bashan et al., 2004)*
	Botryococcus braunii	NO_3^-, TP	(Sydney et al., 2011)*
	Scenedesmus sp.	NO_3^-, TP	(Fierro et al., 2008)*
	Haematococcus pluvialis	NO_3^-, TP	(Kang et al., 2006)*
	Mixed	NH_3, NO_3^-	(Renuka et al., 2013)
	Mixed, *Desmodesmus communis*	TN, PO_4^{3-}	(Samorì et al., 2013)
	Chlorella sp.	NH_3, TP	(Wang et al., 2010)
	Chlorella sp.	TN, TP	(Wang et al., 2013)
Sidestream	*Chlorella sp.*	NH_3, TN, TP	(Li et al., 2011)
	Chlorella sp.	NH_3, TP	(Min et al., 2011)
	Chlorella sp.	NH_3, TP	(Wang et al., 2010)
	Auxenochlorella protothecoides	TN, TP	(Zhou et al., 2012a)

***References as cited elsewhere (Pires et al., 2013)**

The potential benefits of algaculture integration are many, beginning with nutrient removal. All three of the above-mentioned options provide nitrogen and phosphorus removal, which is advantageous over the current practice in many WWTPs (especially in small plants) of focusing on either nitrogen or phosphorus alone. Ecological research is showing that both phosphorus and nitrogen need to be addressed to prevent eutrophication, especially in downstream estuaries and coastal marine environments (Conley et al., 2009). Adding to the benefits, algaculture captures nutrients through cell synthesis instead of through the commonly employed phosphorus removal method of chemical precipitation. Nutrients in algal cell biomass may be more bioavailable than in chemically precipitated sludge solids. However, the degree of nutrient removal benefit will likely vary with the location of the unit process. Side-stream algaculture would likely remove fewer nutrients than primary or tertiary algaculture, simply because it does not deal with the entire wastewater load. It is less predictable whether primary or tertiary algaculture would be advantageous; direct comparisons among the options are needed.

A possible advantage of primary and side-stream algaculture over tertiary is the ability to improve the activated sludge operations. Primary and side-stream processes could remove organic carbon and ammonia, decreasing their levels in the activated sludge influent. Some have reported that the nutrient-rich side-stream centrate is the best stream in a municipal treatment plant for removing nutrients to a high degree while achieving high algal biomass yields (Li et al., 2011; Wang et al., 2010). Combined heterotrophic-photoautotrophic growth has been studied, resulting in greater nutrient removal efficiency, improved lipid yields, and lower algae harvesting costs (Zhou et al., 2012b). This would also decrease oxygen requirements for biological oxygen demand (BOD) removal and nitrification in activated sludge. Additionally, if energy is derived from the algal biomass itself, the decrease in aeration demand could help convert WWTPs from net energy users into net energy producers (Menger-Krug et al., 2012). Further, in the primary and side-stream algaculture scenarios the activated sludge lies downstream of the algal processes where it can deal with any algal biomass that is not separated. These benefits are not available in tertiary algal treatment where there is no feedback stream to the conventional WWTP processes.

Along with nutrient removal algae may impart an improved capability for the removal of hazardous organic contaminants (Muñoz and Guieysse, 2006), and metals (Mehta and Gaur, 2005) though the effects are species and process dependent. It has been shown in some cases that nickel and cobalt have a significant effect on the performance of activated sludge, altering the microbial populations (Gikas, 2008). Algaculture that removes these metals may benefit the overall plant performance. Tertiary treatment would not have an effect here, but primary and/or side-stream algaculture could be advantageous.

With all of the potential benefits, there are certainly hurdles to overcome in integrating algaculture into a WWTP. One main drawback is footprint; because algae utilize sunlight for energy, algaculture reactors are much shallower than other bioreactors (<1 m versus >4 m) and thus much more land area is necessary to achieve the required retention times. This is one of the main reasons to explore algaculture in small treatment systems; small systems are common in rural areas where land is more readily available than in urban areas. Still, minimizing land use is always desirable. This may be one way in which side-stream treatment will be advantageous, with its smaller flow rate and thus smaller reactor size than primary or tertiary treatment.

The cost of new unit processes is always a problem, and certainly for algaculture. In one study of the life cycle costs and environmental impacts for an algal turf scrubber (ATS) treating dairy wastewater, the eutrophication impacts were significantly reduced, but at a cost roughly seven times that of the non-ATS treatment (Higgins and Kendall, 2012). Reducing that cost—perhaps through a synergistic algaculture/WWTP integration—will be necessary to make the ideas feasible.

Other, subtler issues could occur that would be detrimental to an integrated system. For one, activated sludge requires nitrogen and phosphorus to efficiently remove organic carbon from wastewaters. Low nutrient levels can lead to process upsets such as an overabundance of filamentous bacteria or even the production of exocellular slime that severely increases the sludge volume index (SVI), indicating poor settling (Grady et al., 2011). Thus integration of nutrient removal by algae would need to be tailored so as to maintain sufficient nutrient levels in the activated sludge tank. And even if the triacylglycerides (TAG) from algae can be used for biofuel production, it has been reported that harvesting and recycling the nitrogen contained in the non-TAG portion of the cells will be critical to closing the energy balance (Peccia et al., 2013). Advances in biotechnology will likely be needed along with advances in process engineering.

Because the benefits and challenges for algal implementation are complex, the life cycle of the system should be explored to make predictions about the net outcome. Life cycle assessment (LCA) is a systems analysis tool that can be used to identify stages or processes that contribute to a system's overall environmental impacts. LCA is finding increased use for evaluating the sustainability of wastewater treatment plants (Corominas et al., 2013) and can be used to identify potential benefits and impacts of integrating algaculture in wastewater treatment.

This study seeks a fuller understanding of how algaculture can be integrated into small WWTPs. Both process modeling and life cycle modeling are used to explore how this integration may affect treatment operation and the resulting environmental effects, as well as how much algal biomass production may be expected if these technologies are adopted.

Methods

Goal and Scope Definition

The goals of this study are to assess the environmental benefits of using wastewater streams within an existing plant to cultivate algal biomass and to identify potential energy and resource recovery opportunities that algaculture can provide. The focus is on small (less than about 5 million gallon per day [MGD]) WWTPs in the United States.

To ground the study in a realistic scenario, an existing WWTP was chosen as a model: the Cochran Road Wastewater Treatment Plant in Clemson, South Carolina with a service area population of approximately 6,680. It is currently rated at 1.15 MGD with an average flow of 0.6 MGD but there are plans for expansion to 2 MGD in the near future. The existing plant is typical for small systems in rural areas; it is an extended aeration design with an equalization basin, an anoxic selector for control of filamentous bacteria, three aeration basins, two secondary clarifiers, and aerobic sludge digestion. Aerobic digestion is typical at plants this size because it is simpler to operate, whereas anaerobic digestion often requires more advanced training to maintain successful operation. Solids produced from primary sedimentation (primary solids) are problematic for plants without anaerobic digestion, so Cochran Road (like many small plants) does not have primary clarifiers;

through extended aeration, the biodegradable portion of what would be primary solids is treated in the activated sludge aeration basins. Sodium aluminate is added prior to sedimentation for phosphorus removal. Although alum is more common and less expensive than aluminate, the low alkalinity regional water necessitates aluminate over alum.

Expansion of the existing system is being considered in the upgrade. This would include addition of a fourth aeration basin and a third secondary clarifier as well as expansion of the anoxic basin to achieve denitrification through mixed liquor recirculation. In this proposed expansion, efforts to achieve nutrient removal impart large costs to the treatment plant; nitrogen removal will require high energy consumption for aeration (to achieve nitrification) and recirculation pumping (to achieve denitrification), and phosphorus removal will require continued addition of aluminate.

This work models the proposed expanded system (four aeration basins and three clarifiers), but compares the proposed nutrient removal strategy to three types of algaculture integration to achieve nutrient removal. A life cycle approach is used to compare the four nutrient removal strategies with wastewater and algaculture models used to generate inventory data. The functional unit is 2 MGD (7,570 m^3) of raw wastewater treated. There is some debate about the use of raw wastewater as a functional unit for LCAs of such systems due to differences in effluent quality (Corominas et al., 2013); a 2012 study by Godin et al. (2012) recommended the net environmental benefit (NEB) approach to overcome these issues. NEB considers the no action scenario impacts (PI_{NT}) and subtracts from those the impacts from treated wastewater (PI_{TW}) and plant operation (PI_{OP}) to determine the NEB of the processes considered (Equation 2.1). In comparison, a standard LCA would only include the sum of treated wastewater and plant operation impacts (Equation 2.2). The NEB approach is especially useful for wastewater systems because it identifies cross-media effects of treatment, such as the tradeoff between reduced impacts to aquatic ecosystems resulting in impacts to terrestrial ecosystems through land application of biosolids. A modified NEB approach (Equation 2.3) was used in this study to account for these important tradeoffs, while producing results more consistent with standard LCA practices.

$$NEB = PI_{NT} - PI_{TW} - PI_{OP} \qquad \text{(Eq. 2.1)}$$
$$\text{Standard LCA} = PI_{TW} + PI_{OP} \qquad \text{(Eq. 2.2)}$$
$$\text{Modified NEB} = PI_{TW} + PI_{OP} - PI_{NT} \qquad \text{(Eq. 2.3)}$$

The study's system boundaries are drawn at the untreated wastewater leaving the plant headworks (bar screens) and include all emissions to the environment, including effluent discharge, air emissions, and trucking and land application of biosolids. No consideration was given to the impacts from aluminate production, transportation, or disposal. Construction and end-of-life impacts are also outside of the scope.

Treatment Scenarios

The goal of this study was to quantitatively model and evaluate treatment performance and life cycle impacts of several wastewater treatment scenarios, including options with integrated algaculture. The five scenarios considered (Figure 2.1) share the same basic activated sludge and secondary sedimentation systems which serves as a baseline for the rest of the analysis. The four other cases represent modifications to the baseline that are intended to achieve some degree of nutrient removal. The function of all scenarios is to treat two million gallons per day raw wastewater. Each system was modeled using three wastewaters, low, medium, and high strength, as described in Metcalf & Eddy, (Tchobanoglous et al., 2003) to determine the variability in performance.

The baseline system (Base) is the proposed expansion of the extended aeration activated sludge system at the Cochran Road WWTP. This plant is designed to remove BOD and to minimize biosolids production. Nitrification is achieved in this system, converting ammonia nitrogen to nitrate, due to the long solids retention time (SRT, 18 days), but it is not designed to achieve total nitrogen removal by denitrification. Waste sludge is stabilized by aerobic digestion, decanted, and supernatant is returned to the head of the plant.

The second case represents the upgrade proposed to achieve nutrient removal which is commonly used in small systems and is referred to as the conventional nutrient removal (CNR) case. In addition to the baseline system described above, CNR also includes an anoxic tank prior to the aeration tanks, with mixed liquor recirculation, to achieve partial denitrification. Aluminate is added to the mixed liquor prior to clarification to achieve precipitation and thus reduction of phosphorus in the effluent.

The three other systems have integrated algaculture unit processes, each being placed at a different point in the treatment train. The most commonly cited use of algaculture in wastewater treatment is as a tertiary treatment step to remove residual nutrients after activated sludge. This scenario is referred to as tertiary algal nutrient removal (TANR). In another scenario (primary algal nutrient removal, PANR), primary treated effluent is fed to the algaculture system, which serves to remove nutrients prior to activated sludge. This scenario will also require addition of primary sedimentation, which is not common at small treatment plants, to allow light penetration. Finally, side-stream algal nutrient removal (SANR) uses the algaculture unit process to treat concentrated wastewater produced during sludge thickening. This strategy takes advantage of the high nutrient content of the concentrated side stream.

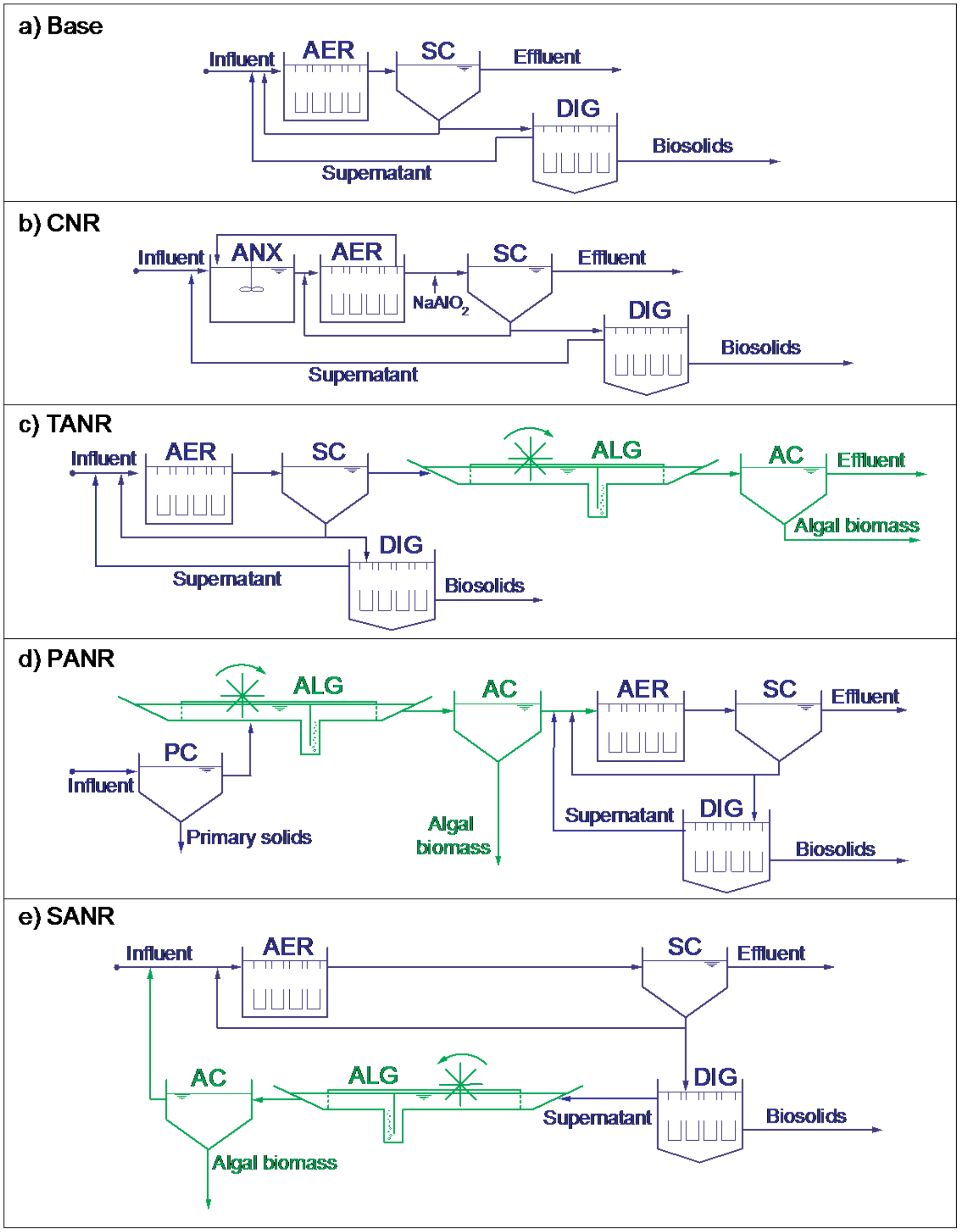


Figure 2.1: Processes and flows for treatment scenarios showing the location of the aeration basins (AER), secondary clarifiers (SC), aerobic digestion (DIG), algaculture ponds (ALG), algal clarifiers (AC), anoxic basin (ANX), and primary clarifier (PC). Processes are: (a) the conventional activated sludge system that serves as a baseline for this analysis, (b) the conventional nutrient removal (CNR), (c) tertiary algal nutrient removal (TANR), (d) primary algal nutrient removal (PANR), and (e) side-stream algal nutrient removal (SANR).

Modeling Approach

For each case, the activated sludge process was modeled using BioWin 4.0 (Envirosim) to determine effluent quality, direct greenhouse gas emissions and biosolids properties for land application. Additionally, algaculture processes were modelled in tandem with Excel (Microsoft) to quantify the changes in aquatic, terrestrial, and atmospheric emissions; the potential algal biomass production; and the land area required for raceways ponds. Algaculture modeling was done using a stochastic approach to evaluate sensitivity (see *Sensitivity Analysis* section); the average output values from algaculture modeling were used as inputs to the BioWin model, where needed. In cases where the two models depended on one another, they were run iteratively until the solutions converged.

The baseline activated sludge model in BioWin consisted of four aerated tanks in parallel, with a total volume of 5.6 ML, a hydraulic residence time of 10.8 hours, and a solids residence time of 18 days followed by three clarifiers in parallel with a combined surface area of 476 m^2. Influent conditions were set *a priori*, except for PANR, for which primary sedimentation and algaculture treatment were modeled and the effluent from these processes served as the influent to the activated sludge system. Side-stream characteristics were determined by the output of the sludge thickening process model in BioWin and from the algaculture treatment model in SANR. BioWin default values were used where not specified. It is recognized that numerical modeling with packages like BioWin has its limitations; models typically require significant parameter verification and comparison with plant data to ensure accuracy. However, for this study the goal is a comparison among process options and by keeping the parameters consistent it is felt that valid comparisons can be made.

Further, there is precedent in the literature for using BioWin models to generate life cycle inventories (Foley et al., 2010); similar methods were used here.

The algaculture process was modeled using nitrogen and phosphorus removals reported in the literature (Table 2.1) and the Redfield ratio (Redfield, 1958) for algal biomass composition ($C_{106}H_{263}O_{110}N_{16}P$). Because these values vary in published reports, and there is inherent uncertainty in how the algae will behave in practice, the modeling input parameters were set as distributions, instead of single values. For each of the three algal-integration scenarios, seven parameter distributions were created: TN and TP removals were the first two, and the stoichiometric coefficients of C, H, O, N, and P were the remaining five. TN and TP removal literature data roughly followed a gamma distribution, so that distribution shape was chosen for modeling. Alpha and beta (shape and rate parameters, respectively) for the gamma distributions were set to best fit the literature data (see supplementary information for more details). Stoichiometric coefficient values for C, H, O, N, and P were generated using normal distributions with the mean of each set to its Redfield ratio value. The standard deviation of these normal distributions was set to 25% of the mean. Each model was run using random numbers within the seven distributions, in a stochastic Monte Carlo approach. Results are reported as the average of 1000 such runs.

A sensitivity analysis was performed to determine which of the seven algae model parameters most affected the results. Each parameter was tested individually, using its distribution in 1000 model runs, but keeping the other parameters set at their mean values. The resulting model outputs for algal biomass production, N uptake into algal biomass, and P uptake into algal biomass were collected as final distributions. The model was considered to be most sensitive to the individual parameters that led to the highest standard deviations in model outputs.

The potential nutrient uptake (removal efficiency multiplied by nutrient loading) for both nitrogen and phosphorus was used to determine the limiting nutrient (N or P) based on the elemental composition of algal biomass. Nutrient uptake was calculated assuming uptake for the limiting nutrient was equal to the potential uptake. Nutrient removal for the non-limiting nutrient was determined by the elemental composition and production of algal biomass. The quality of the effluent was determined based on limiting- and non-limiting nutrient uptake. Nitrogen and phosphorus variables from BioWin that were modeled as available to algae were ammonia, nitrate, readily biodegradable Kjeldahl nitrogen, and orthophosphate. Changes in total organic carbon (TOC) in algaculture were also determined by the elemental composition of the algal biomass, assuming carbon dioxide and TOC were both able to be used as carbon sources for algal growth. Carbon available from wastewater was calculated in BioWin from total dissolved CO_2 and readily and slowly biodegradable COD in the influent to the algaculture process. COD was converted to TOC, as described in Metcalf & Eddy (Tchobanoglous et al., 2003). It was assumed that additional CO_2 would be supplied when CO_2 and TOC in the wastewater were not sufficient to satisfy the demand determined by the elemental composition (i.e. when carbon was the limiting nutrient).

Land area required for algaculture was calculated assuming raceway style ponds as described by others (Park and Craggs, 2010) with a hydraulic residence time of 4 days and a depth of 0.3 m. Dilution of side-stream wastewater is reported in literature and is accounted for in land area calculations. Harvesting efficiency of algal biomass was generously assumed to be 100%, but implications of lower efficiencies are discussed. It is important to note that the purpose of this study is not to design algae ponds for use at treatment plants. Instead it looks at how algaculture could potentially relieve the operational burdens associated with treating oxygen demand and nutrients.

Impact Assessment

A comparative impact assessment was performed and results for the following impact categories are presented: eutrophication, global warming potential, ecotoxicity, and primary energy demand. These categories were chosen to represent the most relevant impacts to treatment operations and emissions. The modified NEB approach was used, where impacts from direct release of untreated wastewater to freshwater were subtracted from operational impacts to determine the net (rather than gross) impacts. The impact assessment is a comparison of the operational stage for the different treatment scenarios; the results are not comprehensive of the entire life cycle of the treatment plant.

This LCA was conducted using GaBi 6.2 (PE International) platform and based on inventory data from process models and the GaBi database for electricity and transportation. Biosolids transportation to agricultural land was modeled assuming 2% solids content and a distance of 100 km from plant to application site in a 22-ton truck. Primary solids generated in the PANR were assumed to be treated off-site and transportation was modeled like biosolids transportation, except 6% solids were assumed because of the better settlability of primary solids (Tchobanoglous et al., 2003). TRACI 2.1 (Bare, 2012, 2011) was the impact assessment method used for eutrophication and global warming. Greenhouse gas emissions were calculated as described previously (Foley et al., 2010). USEtox (Hauschild et al., 2008; Henderson et al., 2011; Rosenbaum et al., 2008) was used for ecotoxicity, which is primarily a result of metals concentrations in biosolids; biosolids metals concentrations were used as reported by Foley et al. (2010). Although considered in biosolids, metals are not reflected in effluent, algal biomass, or avoided emissions which is recognized as a limitation to the calculation of ecotoxicity impacts. Primary energy demand was calculated from United States (East) electricity grid mix and truck transport using GaBi database processes and characterization factors (Professional 2013 and Energy extension databases).

Inventory Results

Analyzing life cycle impacts of a process involves first gathering data on relevant mass and energy flows to build a life cycle inventory. To understand the impacts from an LCA, it is necessary to first interpret the life cycle inventory data to give a better understanding of what is driving the impacts. This interpretation step also allows a better understanding of the drawbacks and potential improvements to the processes analyzed.

Treatment

The primary function of a wastewater treatment plant is to provide a barrier for release of contaminants that will negatively impact the receiving water and thus it is pertinent to understand how new technologies developed for use at wastewater treatment plants will impact effluent quality. Primarily, effluent concentrations of BOD and total suspended solids (TSS) must meet permit limits for discharge (9.5 mg BOD/L and 30 mg TSS/L respectively in the Cochran Road case). For all modeled treatment scenarios, effluent was found to comply with standards for BOD (Table 2.2). In addition, all systems were shown to comply with TSS standards (data not shown). In the TANR case this was directly influenced by the 100% harvesting efficiency assumed for the algaculture process, which is difficult to achieve with current algae technologies(Uduman et al., 2010). In real systems, 100% removal of algal cells would require a robust separation, such as membrane filtration (Babel and

Takizawa, 2010), which would likely impart large energy demands to the algaculture system. Harvesting efficiency and energy consumption of proposed algaculture systems should be addressed prior to implementation of tertiary algal nutrient removal. Implications of harvesting efficiency issues provide motivation for developing an alternative to tertiary treatment for algaculture integration at WWTPs.

Beyond the standard treatment targets of BOD and TSS, effluent nitrogen and phosphorus concentrations are important for controlling eutrophication in receiving waters. Total nitrogen (TN) and total phosphorus (TP) effluent concentrations for each scenario are shown in Table 2.2. All nutrient removal strategies had improved effluent quality in terms of TN over the Base scenario, with TANR and PANR showing the best performance. Again, consideration should be given to the assumption of 100% removal of algal biomass before discharge for the TANR case. For both low and medium strength wastewaters, PANR is also competitive with CNR in terms of phosphorus removal, and has the benefit of non-harvested algal biomass being captured in activated sludge and secondary sedimentation processes.

The effluent quality from SANR is essentially the same as Base; the small flow (approximately 1% of the influent flow) receiving nutrient removal in the SANR scenario does not result in large changes to effluent nutrient concentrations. It should be noted, however, that these results represent a steady-state simulation and side-stream flows are rarely constant, especially for plants that decant digesters as is common for aerobic digesters, such as in the model plant used here. Therefore, the pulse input from the decanting operation could cause a larger perturbation than is captured in this steady-state simulation and thus side-stream algaculture may serve as a type of equalization for small concentrated streams.

Table 2.2: Influent and effluent wastewater characteristics for low, medium, and high strength wastewaters.(G Tchobanoglous et al., 2003) Units are mg/L. The permit limit was 9.5 mg BOD/L for our example treatment plant (Cochran Road); all of the treatment cases were well within that requirement.

	Strength	COD	BOD	TN	TP
Influent	Low	250	122.9	20	4
	Medium	430	211.4	40	7
	High	800	393.3	70	12
Effluent					
Base	Low	20.8	2.6	15.5	2.9
	Medium	30.1	2.6	32.0	5.1
	High	63.5	5.5	54.1	8.5
CNR	Low	19.4	2.2	6.3	0.3
	Medium	28.4	2.2	12.1	0.3
	High	57.8	4.3	20.2	0.8
TANR	Low	16.7	2.6	1.9	1.0
	Medium	24.3	2.6	4.5	1.2
	High	56.9	5.5	9.5	2.2
PANR	Low	17.5	3.2	2.9	0.3
	Medium	19.3	3.2	8.2	0.4
	High	44.4	3.8	16.9	2.6
SANR	Low	20.8	2.6	14.7	3.0
	Medium	30.0	2.6	30.6	5.3
	High	84.6	5.8	52.7	9.2

Reduction of nitrogen and phosphorus from effluent is the result of changing the state of these compounds from the dissolved form to solids or gases. Understanding the fate of nutrients helps elucidate where other impacts occur as a result of nutrient removal. Figure

2.2 tracks the fate of both nitrogen and phosphorus in each case. N and P leaving in biosolids represent the potential benefit of improved soil quality and fertility when biosolids are land applied. However, in CNR much of the phosphorus is bound in stable metal complexes and is not available for plant growth. Additionally, if the end-use of the algal biomass is as a replacement of a terrestrial crop, N and P that leave the plant in algal biomass can also be considered a benefit due to the offsets of fertilizer that would be required to grow the terrestrial crops the algae is replacing.

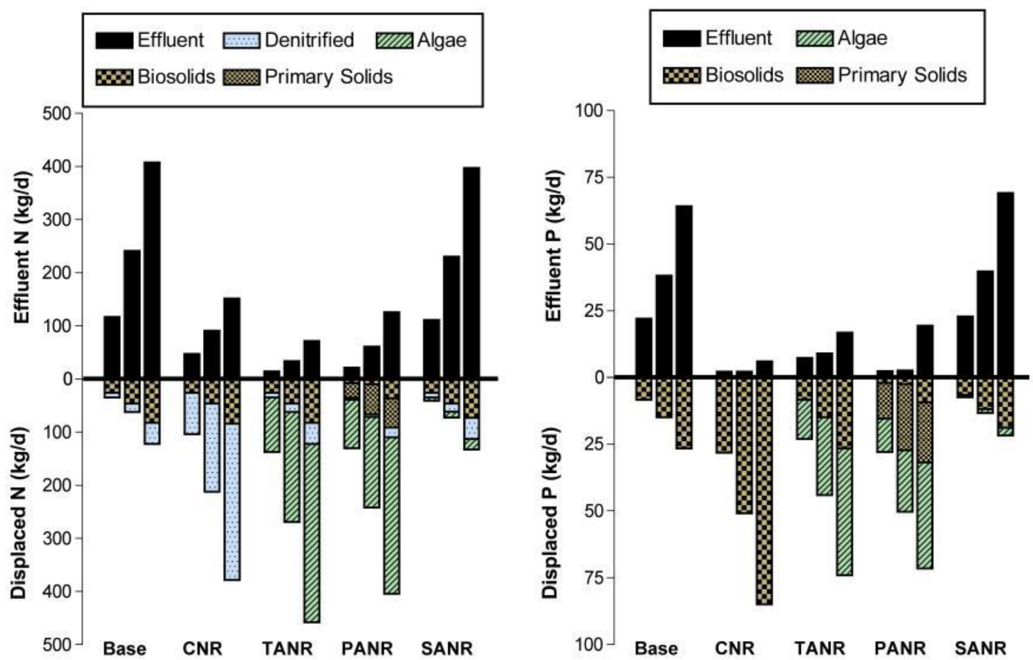


Figure 2.2: Effluent loading and fate of displaced total nitrogen (TN) and total phosphorus (TP) for each scenario. The clusters of three bars for each scenario represent low, medium, and high strength wastewater, respectively.

Nitrogen removal through denitrification (to N_2 gas) is the main approach to nitrogen removal in the wastewater treatment industry, as represented by CNR, but this process is also the main source of nitrous oxide at WWTPs (Kampschreur et al., 2009). This approach to nitrogen removal reduces impacts to receiving waters but because N_2O is such a potent greenhouse gas, may increase overall environmental impacts due to global warming effects, which are discussed in detail later. Implications of primary solids in PANR are also discussed later.

Biosolids Production

Land application of stabilized biosolids is a common method of disposal for small treatment plants and can be viewed as a benefit or an impact to the environmental performance of the plant. On the one hand, nutrients and organic carbon in the biosolids serve to replace industrial fertilizers and sequester carbon by increasing soil organic matter. On the other hand, biosolids have been shown to contain pollutants including heavy metals and other toxic compounds, and land application of these contaminants poses an exposure risk to humans. Additionally, transportation and disposal costs provide incentive to minimize biosolids production. These factors must be weighed in design of plant modifications.

Figure 2.3 shows the results of digested biosolids production from all studied scenarios, including the phosphorus application rate which is the target for nutrient recovery because it is a non-renewable resource. Base, TANR, and SANR cases show similar performance in terms of biosolids production and phosphorus content. CNR resulted in higher biosolids and phosphorus loading rates, but again this can be attributed to the use of chemical precipitation whose metal-bound phosphorus may not contribute well to fertilization of the receiving soil. In addition, the increase in aluminum from aluminate may increase risks associated with land application.

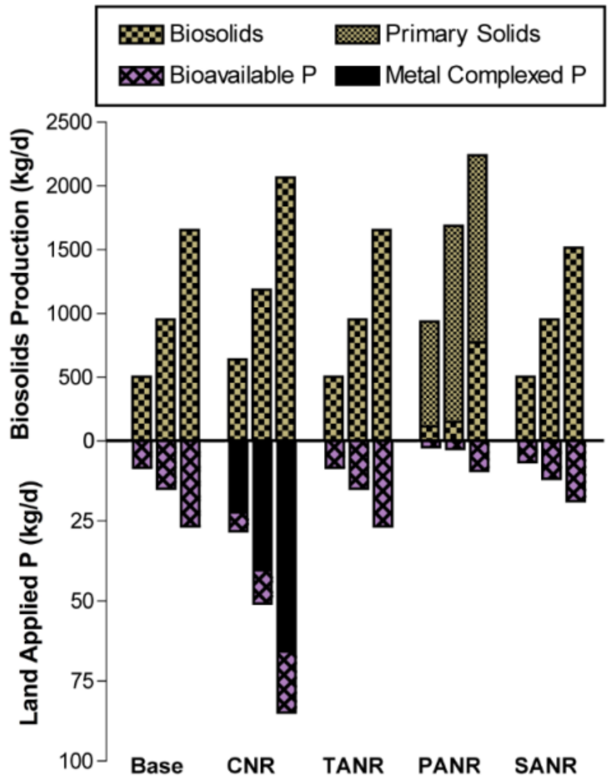


Figure 2.3: Biosolids production rates and phosphorus loading rates to agricultural land resulting from land application . Bar clusters represent low, medium, and high strength wastewater, respectively.

The diminished rate of biosolids production seen for the PANR case is counteracted by primary solids production. Aerobic digestion of primary solids is uncommon, therefore this scenario would only be applicable if an alternative treatment or use of the primary solids is available. Transportation and disposal of the primary solids would be a major consideration for implementation of such a system. One potential end use for the algal biomass could be anaerobic digestion, and if that strategy were employed these additional solids could also be anaerobically digested; this is discussed in more detail later.

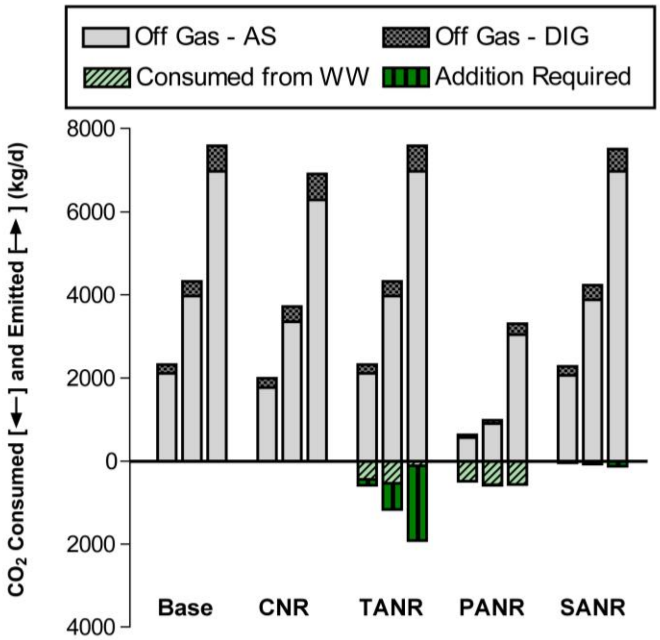


Figure 2.4: Carbon dioxide emissions from activated sludge (AS) and digestion (DIG) and consumption in algaculture, showing both CO_2 consumed from the wastewater and required addition. Bar clusters represent low, medium, and high strength wastewater, respectively.

Direct Greenhouse Gas Emissions

International standards for life cycle assessment state that CO_2 emissions from wastewater treatment are not included in calculations of global warming potential because all the influent carbon is assumed biogenic (Doorn et al., 2006). However, to capture the overall benefits of using algaculture in wastewater treatment, it is pertinent to consider the utilization of carbon dioxide by algae. In the algaculture model, carbon necessary to sustain growth was calculated from the stoichiometric coefficient. Both dissolved CO_2 and readily biodegradable organic carbon in the wastewater were available for algae growth and additional CO_2 necessary was calculated. In both TANR and PANR, it was seen that additional carbon is necessary to achieve the intended nutrient removal due to the lower C:N ratio as compared to untreated wastewater in PANR. This additional carbon requirement could be provided from CO_2 emissions from the activated sludge or digestion processes which produce far more than is required in algaculture (Figure 2.4).

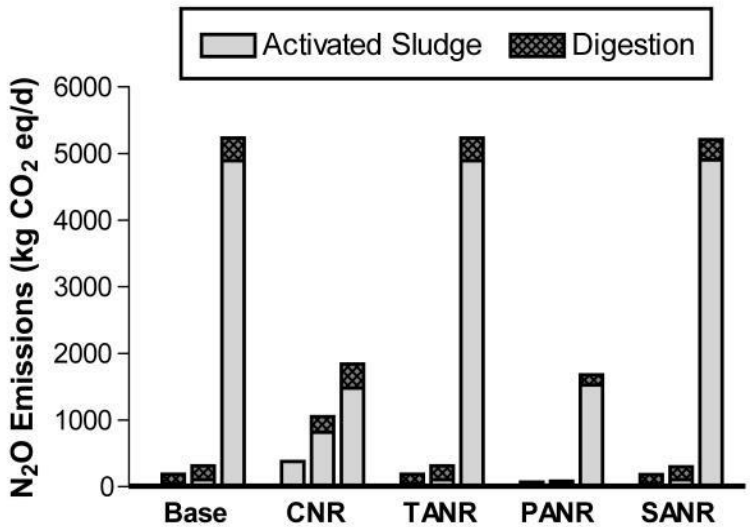


Figure 2.5: Nitrous oxide (N2O) emissions for each wastewater strength (low, medium, and high) showing the influence of high loading rates on global warming potential.

In addition to carbon dioxide, methane and nitrous oxide are potent greenhouse gases that may be produced at wastewater treatment plants. The scenarios considered should not be significant contributors to CH_4 emissions because they do not include anaerobic digestion; this was verified by BioWin models. Nitrogen removal processes (nitrification and denitrification) are often cited as the source of N_2O, but any reactor with low dissolved oxygen can emit this gas. Figure 2.5 shows the calculated N_2O emissions for the activated sludge systems and the digester in each scenario. Though nitrification and denitrification are considered the major source of N_2O, these emissions (in CNR) are minimal when compared to the overloaded systems, except for PANR which was comparable with CNR.

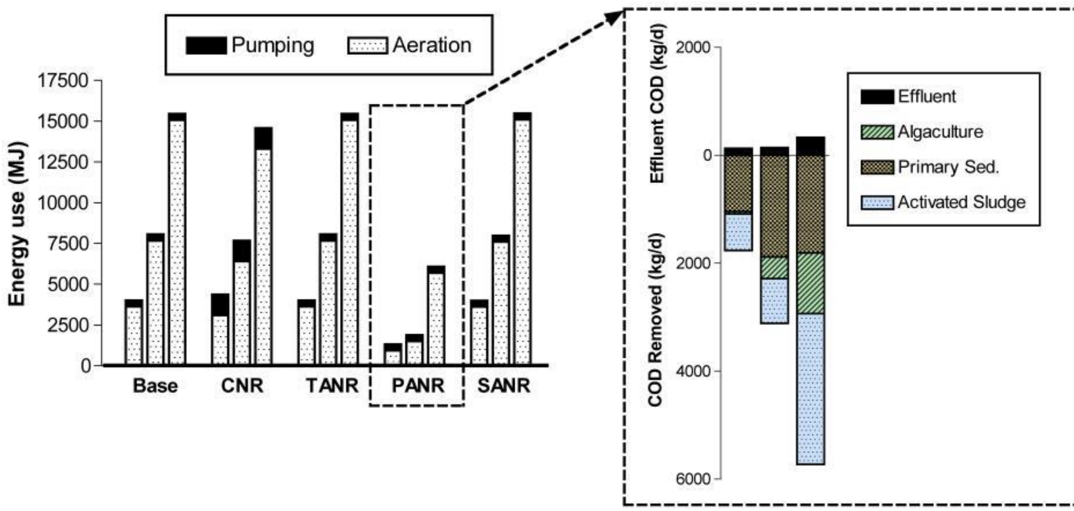


Figure 2.6: Energy use for activated sludge and digestion, showing aeration and pumping contributions (left) and COD removal in each unit operation in PANR (right). Bar clusters represent low, medium, and high strength wastewater, respectively.

Energy Use

Electricity use is a prominent cause of impacts in wastewater treatment life cycle assessment studies. Electricity is primarily used to run blowers to provide aeration to activated sludge systems and for running pumps within the system. Reported aeration rates and recycle pumping rates from BioWin show CNR and PANR reduced the required aeration from the Base scenario (Figure 2.6). For CNR, this is a result of the treatment of BOD occurring in the anoxic selector, which is not aerated. The savings in aeration seen in CNR, however, are the result of recycle pumping required to achieve denitrification in the anoxic selector, thus increasing pumping energy requirements. On the other hand, when algaculture is used prior to activated sludge (PANR), COD loading to activated sludge is reduced, decreasing the aeration requirements for activated sludge. The right panel of Figure 2.6 highlights the influence of primary sedimentation and algaculture on COD removal. In addition to the reduced aeration and recycle pumping rates seen in PANR, it also has the benefit of not requiring additional aeration to algaculture to provide necessary carbon (Figure 2.4) unlike the other algaculture scenarios.

Land Use

The land required for algaculture exceeds that necessary for traditional activated sludge systems due to shallow tank depths necessary to sustain sunlight penetration in algaculture. Results show that for TANR and PANR, approximately 10 hectares are required to support raceway ponds; PANR would also require land for primary sedimentation (approximately 150 m^2 or 0.015 hectares). For SANR, only 0.2 hectares were required, including 50% dilution of side-stream wastewater cited in literature for this type of wastewater.

Sensitivity Analysis

The life cycle inventory for this study relies on predictions about performance for both wastewater treatment unit processes and algal cultivation unit processes. The wastewater treatment aspect is based on BioWin models and, while not perfect, they have been vetted through common use. The algal cultivation modeling is not based on such standard methods and its parameters are less certain. It is therefore interesting to evaluate how sensitive the algae models are to the input parameters.

Sensitivity results for algal biomass production, N uptake into algal biomass, and P uptake into algal biomass are plotted for each algal treatment scenario (TANR, PANR, and SANR) in the supplementary information. The first observation is that algal biomass was more sensitive, in general, to the stoichiometric coefficients for C, H, O, N, and P than it was to the TN and TP uptake parameters. This simply reflects the fact that wider distributions were used for the stoichiometric coefficients than for the uptake parameters.

For predicting algal biomass it will be important to understand the stoichiometric coefficients for the species of interest, under the conditions of interest, in order to limit the prediction error.

The sensitivity results give insight into the behavior of algal unit processes in terms of limiting nutrients. Both nitrogen uptake and phosphorous uptake for the TANR scenario (Figure A7) were sensitive to the N and P coefficients. A closer look at the data (not shown) reveal that during the stochastic TANR modeling N was the limiting nutrient about ¾ of the time while P was limiting for ¼ of the runs. When either nutrient was limiting, it affected both N and P uptake by affecting the total biomass; thus both parameters had an impact on the sensitivity, though N had the greater effect. In the PANR model (Figure A8) P was limiting in 2/3 of the runs, while N was limiting in 1/3 of the runs. This explains why algal biomass and P uptake are most sensitive to the P coefficient, and even N uptake (though most sensitive to the N coefficient) is affected by the P coefficient. In the SANR model (Figure A9) greater than 99% of the runs had N as the limiting nutrient. Thus nitrogen uptake was only sensitive to the TN-uptake parameter, and P uptake was also highly affected by the N coefficient. These results lend motivation for future laboratory and field work to determine which nutrients are limiting in practice, as those will significantly affect the algaculture behavior. Because the wastewater unit processes can dramatically affect the limiting nutrients, and because algaculture can in some cases feed back to the wastewater processes, a clear understanding is needed of how the processes integrate.

Impact Assessment

Life cycle impact assessment is an important tool for engineers, policy makers, and water systems managers for direct comparison of the sustainability of wastewater treatment processes by addressing the tradeoffs between local and global impacts (e.g. eutrophication and global warming, respectively). The impact categories presented in this study were chosen to reflect both primary (at the treatment plant) and secondary (from upstream and downstream processes) impacts of wastewater treatment operation.

The LCA modeling in this study shows both impacts and benefits from treatment operation. Most relevant are eutrophication impacts and benefits (Figure 2.7A). Although there are impacts associated with release of untreated BOD, TN, and TP to receiving waters, use of net impacts shows the huge reductions in eutrophication potential at WWTPs; the magnitude of the benefit directly reflects the effluent quality in each case.

In addition to benefits from reduction of aquatic pollution, there is also a possible benefit in terms of global warming associated with algal nutrient removal (Figure 2.7B). While implementation of TANR may have potential to be a carbon neutral option, the models indicate that PANR is a carbon consuming process within the scope of this study. Treatment and disposal of the primary solids generated in this scenario, which is outside the scope, should also be considered if implementation of this technology is to be sustainable.

Results for both ecotoxicity and primary energy demand assessment show impacts for all scenarios (Figures 2.7C and 2.7D), the lowest in the PANR case. The ecotoxicity and energy demand impacts are consequences of land application of biosolids and electricity consumption at the treatment plant. Ecotoxicity arises from heavy metals which are common, though regulated, in land applied biosolids. The large reduction in biosolids production that results from PANR explains reductions in ecotoxicity for this scenario. Primary energy demand is also greatly affected in the PANR case as a result of several factors. First, aeration required in activated sludge following PANR is far lower due to the

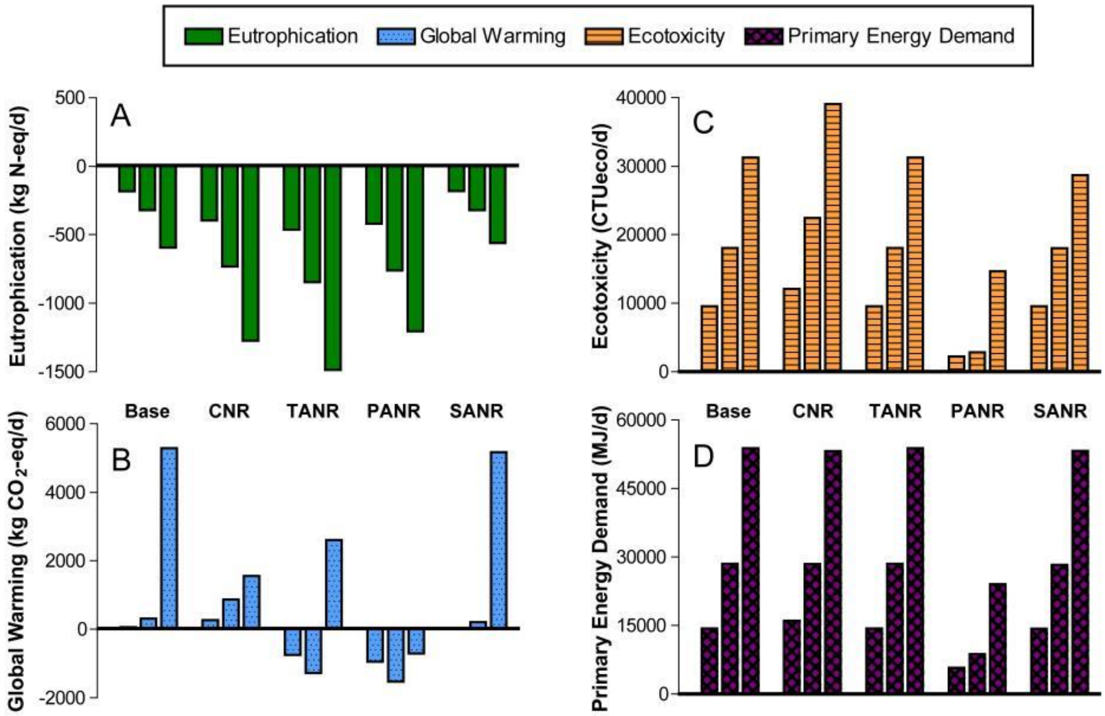


Figure 2.7: LCA results showing eutrophication, global warming, ecotoxicity, and primary energy demand. Negative values reflect a net negative impact, i.e. a benefit. All values are reported for one functional unit (2 MGD of raw wastewater treated). Bar clusters represent low, medium, and high strength wastewater, respectively.

removal of COD by algal growth and primary sedimentation. Additionally, this reduced BOD and nutrient loading to activated sludge is the cause of reduction in biosolids production, which in turn requires less energy for both digestion and transportation to agricultural sites for land application. For a side-by-side comparison of all categories and treatment scenarios, Figure 2.8 shows the impacts on a scale from zero to one, representing the lowest and highest impact respectively in each category; therefore, the smaller a scenario's area, the more beneficial it is. The small size of the PANR petal demonstrates its advantages over the other scenarios. The large relative impact for land use in the PANR scenario identifies one of the drawbacks to this technique, but highlights the motivation for employing the process at small WWTPs, likely in rural areas where land may be more readily available than in urban areas.

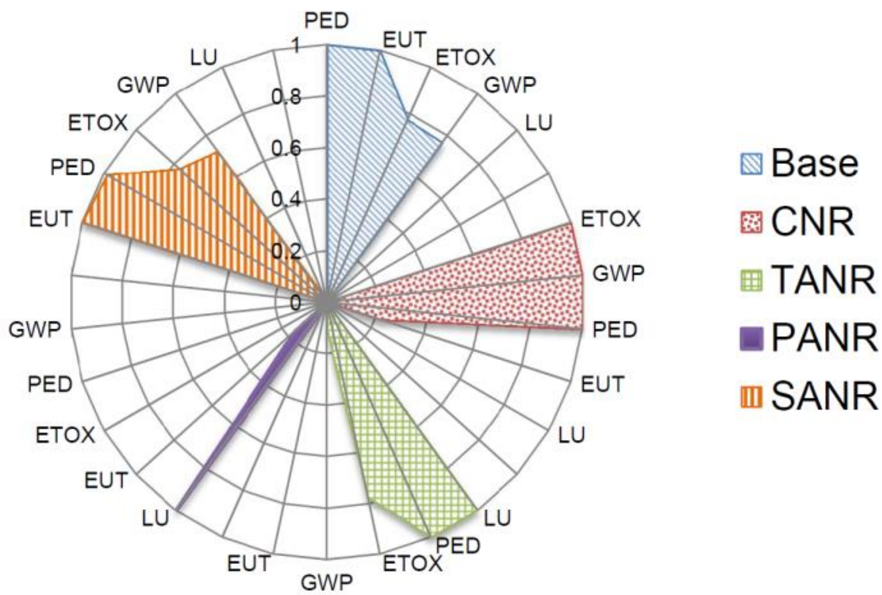


Figure 2.8: Life cycle impacts for the five treatment scenarios in five categories: primary energy demand (PED), eutrophication (EUT), ecotoxicity (ETOX), global warming potential (GWP), and land use (LU). The scale from zero to one represents the lowest and highest impact respectively in each category. Categories for each petal (each scenario) are ordered from highest to lowest impact.

Algal Biomass Production

Comparison of modeled productivities to those reported in literature was used to verify the viability of the modeling approach used; however, previously reported productivities vary greatly, even by an order of magnitude for a given wastewater. In the review by Pittman, et al. (2011) productivities reported for primary treated wastewater (i.e. a TANR scenario) are 26 and 345 mg/L/day, which span the modeled productivities for the three wastewater strengths for TANR in this study (Table 3); a similar trend holds for PANR, where Pittman, et al. (2011) report 25 and 270 mg/L/day, the greater of which required CO_2 addition, which is consistent with the model results reported here. Productivity on centrate (i.e. a SANR scenario) was reported as 2000 mg/L/day, which exceeds any value determined by the algalculture model; however, Zhou, et al. (2012a) reported 269 mg/L/day which is consistent with the model for medium strength wastewater. Additionally, comparison of modeled areal productivities to those reported in literature is informative. Park and Craggs (2010) reviewed algaculture wastewater processes, reporting areal productivities between 12.7 and 35 g/m^2/day. These values are consistent with TANR and PANR with low and medium strength wastewaters, but SANR and all high strength wastewater cases show areal productivities out of this range. This limitation can be explained by the fact that at high nutrient concentrations algal biomass will be too dense for sufficient light penetration which the model does not account for. To be feasible, these systems would require some dilution, thus more land, but would not likely affect other aspects of the treatment process.

Table 2.3: Predicted algal biomass productivity, areal productivity, and methane energy for three algaculture-integrated scenarios for each wastewater strength. Values represent the mean and 95% confidence intervals.

Productivity (mg/L/day)	Low	Medium	High
TANR	56 ± 1	111 ± 2	180 ± 3
PANR	49 ± 1	91 ± 2	156 ± 3
SANR	147 ± 3	267 ± 6	515 ± 12
Areal productivity (g/m2/day)	**Low**	**Medium**	**High**
TANR	16.7 ± 0.3	33.3 ± 0.5	54.1 ± 0.9
PANR	14.6 ± 0.2	27.2 ± 0.5	46.9 ± 0.8
SANR	44.0 ± 1.0	80.1 ± 1.9	154.4 ± 3.6
Methane energy (MJ/d)	**Low**	**Medium**	**High**
TANR	12,170 ± 210	24,100 ± 390	39,140 ± 630
PANR	10,480 ± 170	19,470 ± 330	33,610 ± 570
SANR	680 ± 16	1,270 ± 30	2,360 ± 60

In all ANR scenarios, algal biomass produced could conceivably be used beneficially, either in conjunction with existing treatment operation, or by an outside entity. In the context of the wastewater treatment operation, there are three promising uses. First, land application of algal biomass can provide beneficial nutrients and organic matter to soil. Algal biomass has higher nutrient content than typical biosolids so may be more beneficial as a fertilizer. If land application is chosen, however, it will be pertinent to include the impacts associated with land application, including heavy metals and transportation.

Another option for re-use is as a substrate for anaerobic digestion (AD). Methane energy was estimated using 2 kWh/kg algae (7.2 MJ/kg) as reported elsewhere;(Collet et al., 2011) results are shown in Table 2.3. Although AD is not common for small plants, it has been proposed that a centrally located site for anaerobic digestion may serve to digest neighboring systems' biosolids (Qi et al., 2013). It is also recommended that accepting other organic wastes can improve payback periods for digesters. If ANR can serve as a substrate for biogas production and as a means to decrease costs associated with wastewater treatment, this may further improve payback periods.

In addition to land application and biogas production, algal biomass from nutrient removal processes could serve another wastewater treatment purpose as a biosorbant. Algae have been shown to be effective in removal of metals and other contaminants present in wastewaters at low concentrations, and could potentially be used on site at municipal WWTPs or distributed for use at contamination point-sources. These point sources would likely be factories or other industrial wastewater producers.

Recommendations

Treatment, algaculture, and life cycle assessment models in this study have shown the benefits of using algal nutrient removal at small wastewater treatment plants, but further laboratory and pilot scale research is necessary to move this technology into the real world.

Wastewater specific algal growth rates, nutrient uptake rates, and areal productivity values will be necessary to design functional ANR systems. Improved algaculture models should also be pursued allowing for optimization of integrated processes.

Conclusions

This study supports the hypothesis that integrating algaculture at wastewater treatment plants can improve the sustainability of wastewater systems. Primary algal nutrient removal proved most promising due to huge reductions in operational energy and biosolids production. However, this scenario would require primary sedimentation, which is an important consideration. Improvements in effluent quality and efficiency over conventional treatment strategies through algal nutrient removal can provide an innovative way for small communities to contribute to a growing interest in energy and resource recovery in the wastewater industry.

CHAPTER THREE

EFFECTS OF NITROGEN LIMITATION AND CULTURE DENSITY IN ALGAE SYSTEMS USING MICROFILTRATION

Abstract

Membrane photo-bioreactors have been proposed for use in algae-based wastewater as a promising technique to achieve simultaneous nutrient removal and pre-harvesting of algal biomass. However, there is limited research currently available that informs how the relationship between culture conditions (nutrient and biomass concentrations, for example) and membrane fouling will affect these systems. This work made use of bioreactors with *Synechocystis sp.* operated for 107 days at various nitrogen and biomass concentrations to investigate this relationship. Nitrogen limitation was found to exacerbate membrane fouling. The proposed mechanism for increased fouling is accumulation of carbon-rich intracellular metabolites and subsequent diffusion from the cell. These results can be used to inform design and operation of membrane-based algal wastewater treatment or biofuels applications.

Introduction

The potential benefits of combined algaculture wastewater treatment (WWT) systems are well established; a number of review articles have focused on this symbiotic relationship (Christenson and Sims, 2011; Olguín, 2012; Pittman et al., 2011; Rawat et al., 2011; Razzak et al., 2013). Briefly, algae can potentially provide nutrient removal at WWT plants with lower energy use, and thus cost, than conventional biological nutrient removal (BNR) WWT. Meanwhile the biofuels and biomanufacturing industries, in which algal biomass is potentially valuable, would benefit from free nutrients and freshwater for production of this resource.

Despite these advantages, there are shortcomings associated with using algae in WWT. The open ponds which have historically been the focus of algaculture WWT processes (because of the lower capital and operating costs when compared to closed photo-bioreactor (PBR) systems) are limited by poor light utilization, inadequate diffusion of carbon dioxide from the air, large land areas required, and contamination by problematic organisms (Razzak et al., 2013). Additionally, control over the growth conditions in open ponds is low (Christenson and Sims, 2011). Operational difficulties with open systems have motivated an increased interest in improving operational costs of closed PBR systems as well as the ability to use them on a large scale.

Membrane photo-bioreactors (MPBRs) , which are PBR systems that use membrane filtration for biomass/growth medium separation, have proven effective for providing nutrient removal when combined with conventional treatment trains (Gao et al., 2014, 2015; Honda et al., 2012) and activated sludge membrane bioreactors (MBRs) (Marbelia et al., 2014; Singh and Thomas, 2012) and may provide a practical solution to many of the issues with current algal WWT systems. Complete biomass retention via membrane separation minimizes washout and enables decoupled hydraulic and solids retention times (HRT and SRT) and improved control over dilution rate and biomass concentrations (Bilad et al., 2014; Gao et al., 2014). Through higher biomass concentrations, volumetric productivities and nutrient removal efficiency can be improved, allowing for smaller footprint (Gao et al., 2015; Honda et al., 2012; Marbelia et al., 2014). Membranes provide more effective harvesting without the large land area required for gravity sedimentation and without any chemical additions (e.g. flocculants) which can impact both biomass and effluent quality (Gerardo et al., 2014). The resulting algal biomass product is also more concentrated than in a typical PBR, reducing harvesting requirements for industrial applications (Pavez et al., 2015).

Nevertheless, membrane fouling is a major issue that increases operational costs of running membrane systems. Many studies have investigated the dominant fouling mechanisms by algal cultures. While hydrophilic membranes have been shown to reduce fouling tendency over hydrophobic membranes (Sun et al., 2013), the composition of the algal culture is also an important factor in fouling behavior. Algae cells, bacteria and other microorganisms, and other organic components all foul membranes to some extent and the contribution of particular fractions is dependent on pore size and culture conditions. One study (Rickman et al., 2012), which examined one ultrafiltration (UF) and two microfiltration (MF) membranes (50 kg/mol, 0.22 μm, and 5 mm pore sizes, respectively) found that most fouling components passed through the large pore size and were retained on intermediate pore size (0.22 μm) membranes; thus whole algal cells which were retained by the 5 mm membrane were not found to be a major foulant. However, when algal cells complex with other organic material, a compressible cake is formed causing significant declines in flux (Babel and Takizawa, 2010; Li et al., 2014; Zhang et al., 2013). Bacteria or other organisms, algal cell fragments, and other microparticles, are all shown to contribute to high-resistance cake layers (Li et al., 2014; Pavez et al., 2015; Rickman et al., 2012). These components are often the result of non-axenic cultures, decay, and shear stress from processes such as pumping or mixing (Ladner et al., 2010; Wicaksana et al., 2012), all of which would be expected in an algal WWT context. Dissolved algogenic organic matter (AOM) has also been shown to play a large part in fouling. Results vary between studies in whether large

(Pramanik et al., 2015) or small (Zhang et al., 2013) AOM molecules are more problematic. This discrepancy may be explained by differences in pore size used in the study or the composition of the AOM, which are species-specific and may vary under different culture conditions (Zhang et al., 2013), as small hydrophobic molecules of AOM may be adsorbed inside of pores causing a reduction in pore size and membrane flux (Li et al., 2014).

From a WWT perspective, it is important to understand how culture conditions might affect fouling, and thus performance, of an MPBR. Culture age has a large influence on the composition of AOM. Extracellular organic matter (EOM) is mainly produced during exponential growth (Myklestad, 1995; Nguyen et al., 2005), whereas cellular organic matter (COM) is released as cells age and begin to decay (Pivokonsky et al., 2014). EOM is typically low molecular weight (MW) intermediate products of metabolism that diffuse through cell membranes due to high concentrations within the cell (Nguyen et al., 2005); protein is accumulated during log phase growth but polysaccharide production becomes dominant as nitrogen and phosphorus are depleted from the growth medium (Myklestad, 1995). Dilution of algal cultures can lead to higher EOM production per unit biomass, as the diffusion process is driven by the intra- versus extracellular concentration gradient. COM generally consists of higher MW compounds and is composed of proteins and polysaccharides that result from the degradation of cell material (Nguyen et al., 2005; Pivokonsky et al., 2014)

In a WWT-MPBR system where culture density is determined by growth rate and biomass harvesting (or wastage) and the ultimate goal is to deplete nutrients in the growth medium/wastewater, it is expected that these factors will affect the production of AOM and consequently fouling behavior; however, these effects have not been adequately addressed in the literature. There are many studies that discuss MPBRs in a nutrient removal context

(Gao et al., 2015, 2014; Marbelia et al., 2014; Singh and Thomas, 2012), or from a fouling perspective (Babel and Takizawa, 2010; Huang et al., 2010; Pramanik et al., 2015; Rickman et al., 2012; Zhang et al., 2013), but these studies did not treat both together. One study (Bilad et al., 2014) addressed nutrient removal and discussed fouling briefly, but did not investigate the possible relationship between the two. Therefore, this work aims to explore the relationship between nutrient condition, culture density, and fouling behavior during membrane filtration. It is hypothesized that nitrogen limitation results in production of extracellular organic matter that promotes fouling during membrane filtration.

Methods

Algal Culture

All experiments were performed with non-axenic cultures of the unicellular cyanobacteria *Synechocystis sp.* which was maintained by serial inoculation (10% v/v) into autoclaved BG-11 medium in 1L media bottles every two week. Cultures were aerated with 300 mL/min humidified compressed air and grown under 12 hour light cycles using fluorescent lighting with an intensity of approximately 5,000 lux measured at the surface of the bottles using a digital light meter. Before inoculation the cells were washed three times by centrifuging the culture, discarding the supernatent, and re-suspending the pellet in 0.01 M bicarbonate buffer; the pellet was re-suspended in the appropriate feed media (described below) after the last centrifugation and added to the experimental bioreactors.

The three semi-continuous bioreactors used in the study had a culture volume of 750 mL and a feed rate of 250 mL/day (an HRT of 3 days). Both starting media and feeds were based on BG-11 medium with modified nitrogen content (5, 10, and 20 mg N/L sodium nitrate) for low, medium, and high nitrogen treatments to simulate a range of concentrations expected for secondary treated wastewater where algaculture might be used as tertiary treatment. Media were prepared weekly and autoclaved in media bottles then re-suspended with distilled, deionized water to the original mass to account for evaporation during autoclaving and to maintain constant feed nitrogen concentrations.

The bioreactors were operated for approximately 15 weeks (107 days) with three separate stages (Table 3.1). The daily maintenance and monitoring routine for all stages included culture density and biomass measurements, fouling experiments, and bioreactor feeding (full data sets for culture density and biomass can be found in Appendix B). In the first stage, the bioreactors were inoculated as described above and biomass was allowed to grow and acclimate to the culture conditions; in this stage only a small amount of the culture was removed daily (50 mL) for culture density (based on optical density, described below) and biomass measurements, and was replaced with permeate from fouling experiments (described below) to maintain a total volume of 750 mL in each bioreactor.

The second stage began when the culture density in all bioreactors began to level off. During the second stage the biomass density was normalized between the bioreactors daily. Density was recorded at the beginning and end of each daily routine to confirm densities were normalized correctly. For the first 30 days of the second stage additional portions of whole culture were removed (later referred to as harvesting) and replaced by permeate from fouling experiments to normalize the cultures to the minimum culture density of the three treatments. During the latter half of the second stage, all cultures were maintained at a constant density for at least one week before changing the target density.

Prior to the third stage, each bioreactor was exposed to a single nitrogen concentration (20, 10, and 5 mg N/L for High-N, Medium-N, and Low-N bioreactors, respectively) and observations regarding the impacts of nitrogen limitation on filtration performance were made. Feed nitrogen concentrations were adjusted for the third stage in order to test the response of each reactor. During this stage, the High-N and Medium-N cultures, which did not experience nitrogen starvation in the first two stages, began receiving 5 mg N/L feed; the Low-N culture, which was nitrogen limited during the first two stages, began receiving 20 mg N/L feed. Cultures were then allowed to acclimate again for one week while normalizing to the minimum density of the three cultures. Finally, the cultures were maintained at a constant density for the last week of operation. During this stage, the cultures were referred to as H-L (High- to Low-N), M-L (Medium- to Low-N), and L-H (Low- to High-N) to indicate the nitrogen concentrations each reactor experienced during stage one/two and stage three, respectively.

Table 3.1: Bioreactor timeline summary

Bioreactor Stage	Biomass Density Target	Length
Biomass growth and acclimation	Biomass density not adjusted	35 days
Normalized biomass density	Lowest OD_{595} (30 days) Constant OD_{595} (0.40, 0.35, and 0.30; $\geq$ 7 days each)	58 days
Changed N regime	Lowest OD_{595} (7 days) Constant OD_{595} (0.35; 7 days)	14 days

Daily Maintenance and Monitoring

Prior to daily maintenance and monitoring, cultures were re-suspended to original mass with distilled deionized water daily before samples were taken to account for any evaporation that occurred. Culture density was measured as optical density (OD_{595}) through absorbance measurements at 595 nm; samples of each culture were dispensed into 8 wells (300 µL each) of a 96-well plate, and absorbance was recorded. On days when biomass was harvested (to achieve a desired OD_{595}), 1:3 and 2:3 dilutions of each sample were also measured and the amount of culture to be replaced by membrane permeate was determined based on the linear fit of OD_{595} versus dilution data. On harvesting days, an OD_{595} reading was also recorded after feeding the bioreactors.

Biomass was measured as total suspended solids (TSS) [i.e. total solids (TS) minus total dissolved solids (TDS)]. TS and TDS were determined by weighing the mass of solids remaining after triplicate 15 mL aliquots of whole culture and membrane permeate were dried in a 105 °C oven for 48 hours.

Fouling Experiments

Membrane filtration experiments were carried out using cellulose acetate flat sheet membranes with 0.22 μm nominal pore size, cut into 63 mm diameter round coupons for use in a dead-end Amicon cell. Before filtration of algal cultures, the membranes were seated using distilled (DI) water under 10 psi of pressure until 500 mL had passed the membrane; then an additional 500 mL clean water flux (CWF) filtration was run at 4 psi after which the pressure was maintained for the rest of the fouling experiment.

Fouling tests were performed using 95 mL of algae culture poured directly into the Amicon cell, so as to retain all biomass on the membrane coupon to be returned to the bioreactors. Flux curves were determined using permeate mass readings recorded in LabView once every second. After the culture had drained from the Amicon cell, the unit was disconnected and the biomass was rinsed from the membrane with 5 mL of feed three times and biomass/feed concentrate was collected to be returned to the bioreactor during the feeding procedure. Permeate samples were collected and stored in the refrigerator for dissolved organic carbon (DOC) and total nitrogen (TN) analysis (Shimadzu TOC-VCPN) (full data sets for DOC and TN can be found in Appendix B). The membrane was then rinsed three times with DI and returned to the Amicon cell. Duplicate culture filtrations were run every day on a single membrane coupon for each treatment. Recovery flux (RF) as determined after the second culture filtration and after the membrane was rinsed as described previously using DI water. Membrane coupons were stored in DI water overnight in the refrigerator and used for at least one week before being replaced.

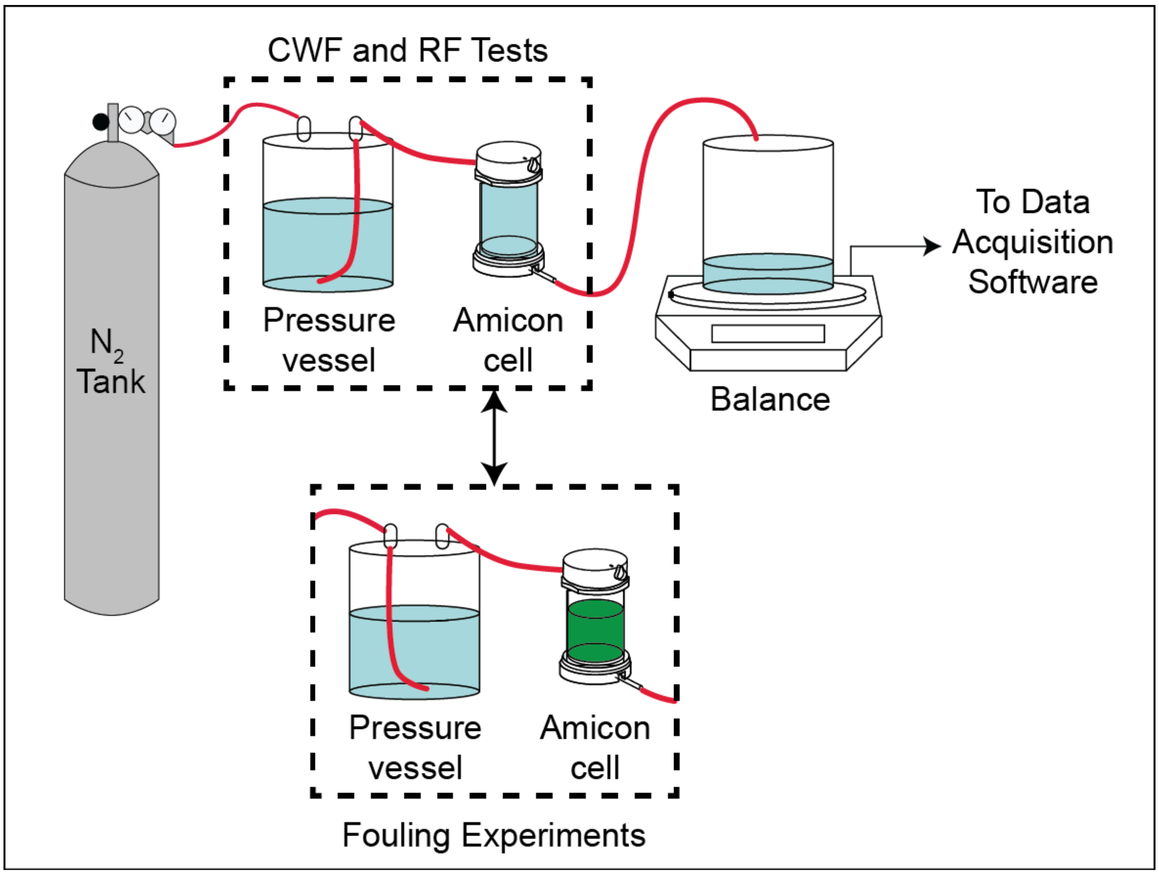


Figure 3.1: Laboratory filtration setup

Bioreactors were fed after completion of the fouling experiments. Concentrated culture, collected when washing the membrane coupons after filtration, and remaining feed solution were returned to the bioreactor. Then the bioreactor was weighed and permeate was used to bring it to its original mass in order to account for minor losses during filtration.

Results

Fouling for High-, Medium-, and Low-N cultures, maintained at a constant density for one week, were compared; densities corresponded to OD_{595} values of 0.40, 0.35, and 0.30. For all three culture densities considered, the Low-N treatment consistently had the most severe fouling, as seen by the long filtration times during fouling experiments (Figure 3.2). However, the High-N treatment also had a high propensity for fouling at the highest density (OD_{595} = 0.40) considered. Both the High-N and the Medium-N treatments showed less fouling at the lowest density (OD_{595} = 0.30) and the Low-N treatment's fouling was also less prominent after three days at that density.

Fouling that was not able to be removed by the rinsing procedure was observed during the recovery flux (RF) tests after duplicate culture filtrations each day, and by subsequent days' clean water flux (CWF) tests. CWF and RF were normalized to the coupon's original CWF (Figure 3.3). The lowest density cultures had the least decline in CWF and RF for all nitrogen treatments. The High-N treatment showed the most severe fouling at the highest density; additionally, the High-N CWFs were consistently higher than the previous day's RF. At the median density observed, all nitrogen treatments showed significant declines in CWFs and differences between the CWF and the previous day's RF.

Table 3.2: Biomass concentrations, in mg TSS/L, at three optical densities (OD_{595}).

	0.40	0.35	0.30
High-N	281 ± 15	249 ± 26	202 ± 15
Medium-N	287 ± 8	242 ± 12	225 ± 13
Low-N	307 ± 11	260 ± 20	251 ± 84

Over the duration of these experiments, biomass concentrations, measured as (TSS), were similar for all nitrogen treatments and decreased by 14.1 ± 0.1% between the 0.40 to 0.35 step and 9.6 ± 0.8% between the 0.35 to 0.30 step for all treatments (Table 3.). No trend was observed for changes in permeate TN or dissolved organic carbon DOC as a result of the change in density. Permeate TN concentrations were 10.54 ± 0.52, 1.69 ± 0.35, and 0.16 ± 0.02 mg N/L and DOC concentrations were 1.54 ± 0.13, 1.56 ± 0.16, and 1.83 ±0.15 mg DOC/L for the High-, Medium-, and Low-N treatments, respectively, averaged over all densities. Qualitative differences were also observed between the bioreactors; the Low-N culture was the palest in color, followed by Medium-N, and was attributed to "bleaching" often associated with nitrogen starvation. Individual cells were also larger in the Low-N culture when observed microscopically.

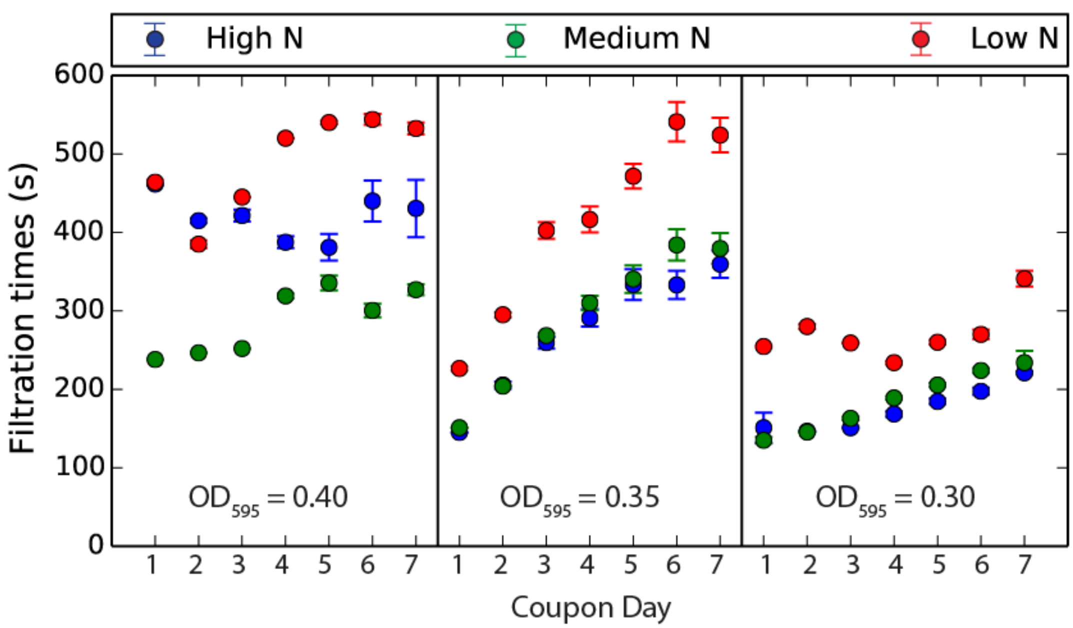


Figure 3.2: Time to filter 90 mL of High-, Medium-, and Low-N cultures at optical densities of 0.40, 0.35, and 0.30. CWFs range from 300 – 400 LMH. Markers represent average of two replicates and error bars show range.

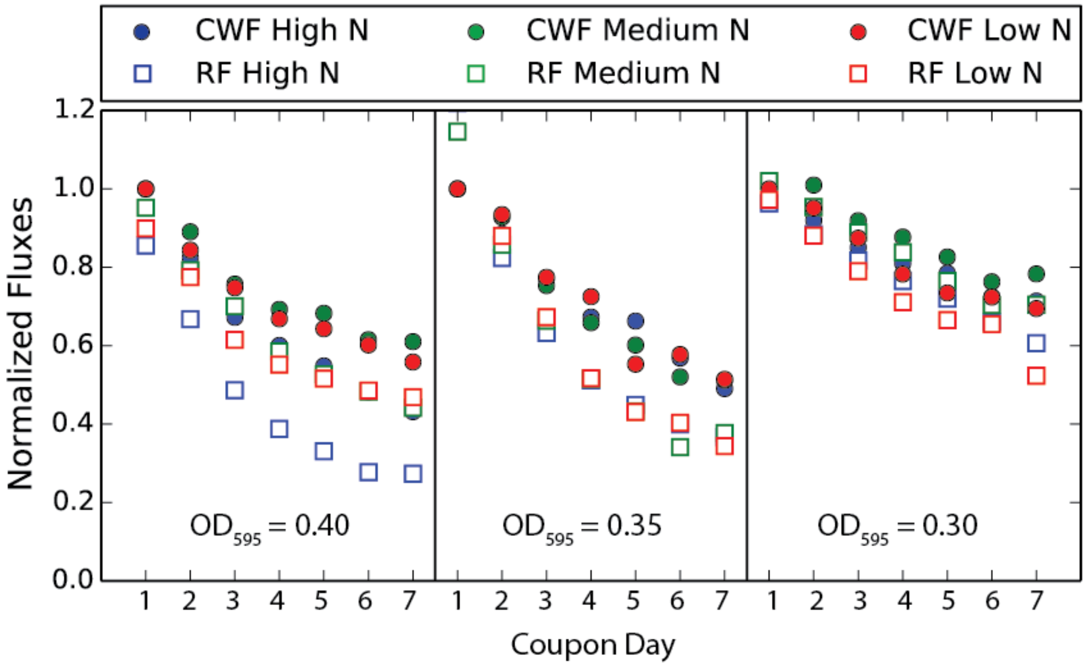


Figure 3.3: Daily CWF and RF (normalized to coupon's initial CWF) for three membrane coupons when cultures were maintained at optical densities (OD_{595}) of 0.40, 0.35, and 0.30 for one week. CWFs range from 300 – 400 LMH.

The impacts of changing the feed nitrogen concentrations for all reactors were also observed; the modification period was compared to periods before and after the transition with a similar density (OD_{595} = 0.35) (Figure 3.4). The biomass density originally dropped in all bioreactors immediately following the change in feed and then slowly increased again. TN also dropped quickly after the switch, falling below 1 mg N/L in 4 days for the H-L reactor and only 1 day for the M-L reactor. The TN rose in L-H reactor, reaching 14 mg N/L after two weeks on switched feed. DOC increased in the second week after the switched feeds.

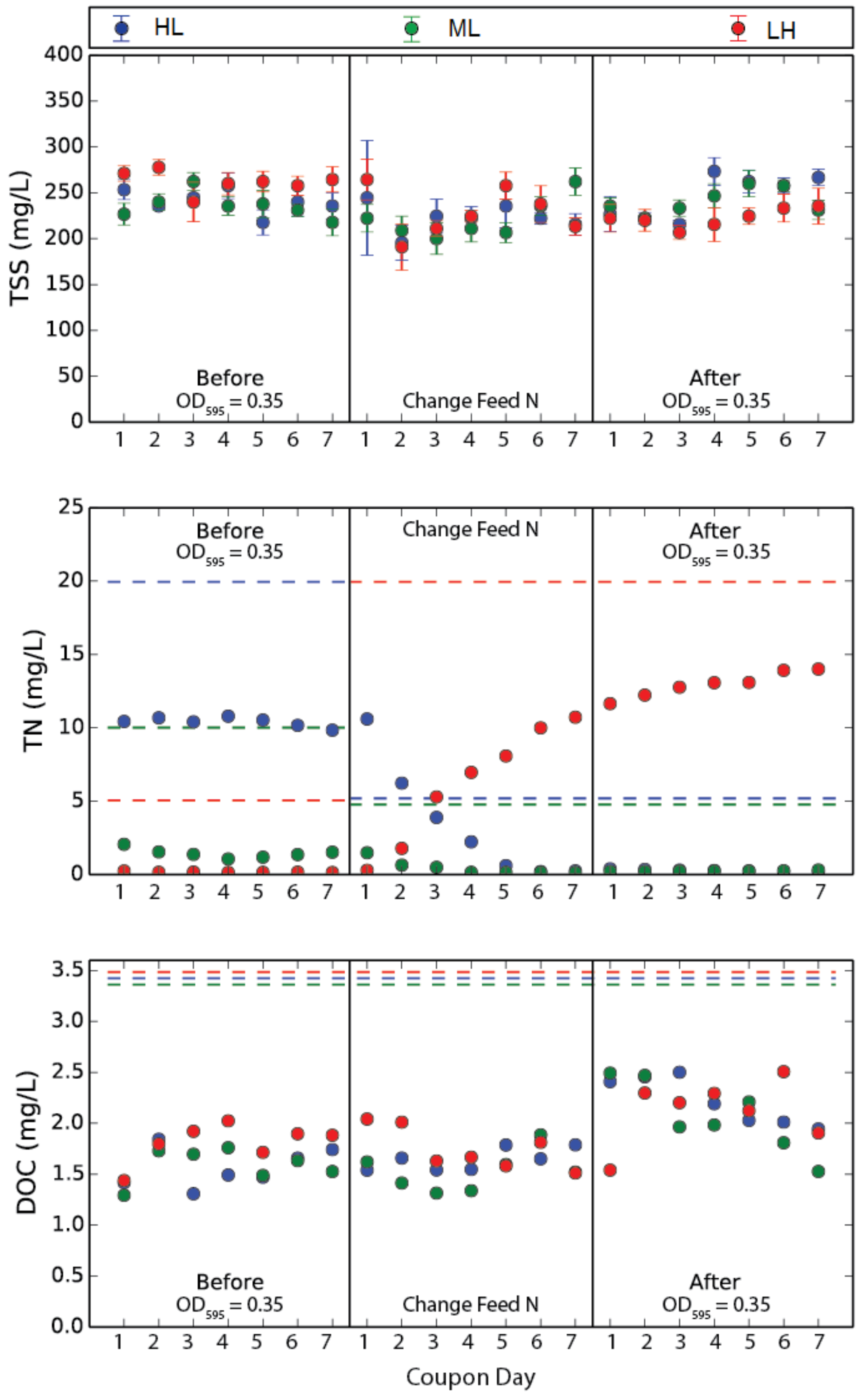


Figure 3.4: Total suspended solids (top), total nitrogen (middle), and dissolved organic carbon (bottom) before, during, and after feed media N change. The bioreactors starting receiving switched feed the first day of the second week (panel labeled "Change Feed N"). Dashed lines in TN and DOC plots indicate feed concentrations.

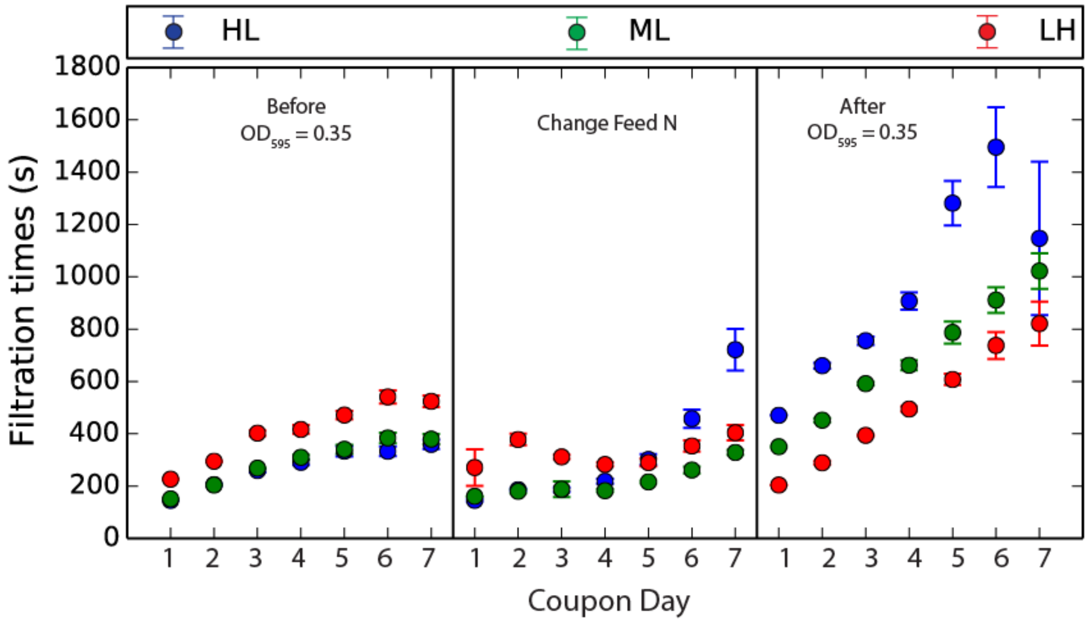


Figure 3.5: Time required to filter 90 mL before, during, and after the transition in N concentrations of feed media. CWFs range from 300 – 400 LMH. Markers represent average of two replicates and error bars show range.

Prior to the change in nitrogen concentration, the L-H reactor showed the most fouling, as observed previously; however, for three days thereafter the flux performance gradually improved until the filtration times of the H-L reactor was highest for the last two days in the week of the transition (Figure 3.5). All cultures showed a high degree of fouling in the final week. A significant amount of irreversible fouling was observed, as seen by low initial fluxes observed throughout the week.

Discussion and Conclusions

This work supports the hypothesis that nitrogen limitation promotes extracellular organic matter production which causes fouling in algae culture/microfiltration systems. More fouling was generally observed when nitrogen was limited in the algal cultures, as was the case for the "Low-N" reactor for the majority of fouling experiments performed. This hypothesis was also upheld when, following the switch from 20 mg N/L feed to 5 mg N/L feed, the "High-N" reactor showed more fouling than previously observed. However, the fouling seen for the "Medium-N" reactor did not immediately confirm the hypothesis. Despite having depleted the TN in this reactor in a single day after switching feeds (before the filtration at 400 mL in the second panel of Figure 3.5), it continued to maintain the highest flux of the three reactors for a week.

These results suggest that the relationship between fouling and nitrogen limitation is complex. When algal cells experience prolonged nitrogen stress, they begin to accumulate carbohydrates or lipids, which is a technique often utilized in the algal biofuels industry (Lardon et al., 2009). The concentration gradient between the inside of the algal cells and the surrounding growth medium could cause diffusion of metabolites, or EOM, which causes fouling. The exacerbated fouling observed between OD = 0.40 and 0.35 (Figure 3.2) may also be caused by this, as dilution of the biomass would exaggerate the intra- versus extracellular gradient of metabolites (Nguyen et al., 2005). The foulants observed at these densities that were not able to be removed with DI rinsing were likely hydrophilic in nature, as explained by the increase in CWF from the previous day's RF, due to foulant removal by dissolution in DI as the membrane is soaked overnight.

Finally, the nitrogen stress mechanism may also explain why the Low-N reactor had a higher biomass concentration than the other treatments despite having normalized the densities using optical density measurements; by accumulating carbon within the cell, individual cells in the Low-N reactor contained more mass, despite the replication rate of the culture being lower than in a nutrient rich environment (Lardon et al., 2009).

There are several possible explanations as to the cause of the sizeable increase in fouling in the week following the media switch. First, as previously noted, the replication rate of the High-N cultures was much greater than for the Low-N as it was not nitrogen limited. Therefore, when the transition to lower nitrogen feed occurred and the cells accumulated carbon-rich metabolites, there was a greater number of cells to release EOM and cause fouling. Additionally, it is possible that after 100 days, the cultures began to decay and the release of COM was the major cause of the fouling observed. This corresponded with a rise in permeate DOC (Figure 3.4) which suggests that there was an increase in release of small MW molecules which can pass the 0.22 μm pore-size membrane.

This study supports the hypothesis that nitrogen limitation and culture density impact membrane performance during algal filtrations. Accumulation of intracellular carbon-rich molecules due to nitrogen stress and subsequent diffusion of these molecules out of the cells is the proposed mechanism for this effect. This mechanism should be tested experimentally by tracking the fate of the metabolic products of carbon fixation; one option for doing so is to use radio-labeled bicarbonate in the growth media. Thereby the accumulation of intracellular carbon-rich compounds can be experimentally observed, as well as the dissolution of these compounds into surrounding medium. If this approach further confirms the hypothesis, it is suggested that chemical composition of the fouling-related compounds be studied to help identify the biochemical basis for the effect. In doing so, strategies can be developed to mitigate fouling in membrane-based algaculture systems.

The impact of the relationship of culture conditions and fouling propensity have implications if membrane filtration is to be used for algal WWT or biofuels applications. In both cases, nitrogen starvation is desirable (in WWT for effluent quality purposes and in biofuels to stimulate the accumulation of energy-rich metabolites) but may impose operational burdens as a result of increased fouling. Therefore, these are important considerations for implementation of membrane-based algaculture systems.

CHAPTER FOUR

COMPARATIVE LIFE CYCLE ASSESSMENT OF NUTRIENT REMOVAL OPTIONS FOR EXISTING LAGOON SYSTEMS: ATTACHED GROWTH ALGAE RETROFIT VERSUS GREENFIELD ACTIVATED SLUDGE CONSTRUCTION

Abstract

Lagoon systems were once a common method of wastewater treatment, but have fallen out of favor due to their large land requirements and unreliable ammonia control. Activated sludge systems utilizing biological nutrient removal (BNR) and/or chemical nutrient removal are often built to replace lagoon systems when more stringent nutrient limits are set. Alternatively, if existing lagoons could be upgraded to achieve better nitrogen and phosphorus removal, this approach may be more environmentally friendly than greenfield construction of activated sludge systems. Pilot studies at an existing lagoon system in Logan, Utah indicate that utilizing rotating algal biofilm reactors (RABRs) in conjunction with lagoons can provide nutrient removal to meet tightening permit requirements. This study aimed to compare two upgrade approaches (BNR activated sludge and RABR installation) using life cycle assessment to determine the tradeoffs for each system. Historical data from the Logan lagoons, pilot data of the RABR systems, and activated sludge modeling using BioWin were employed to generate a life cycle inventory. Eutrophication, global warming potential, and cumulative energy demand were the impact categories considered. Results show that the lagoon with RABR system improved eutrophication impacts by 85% relative to the existing lagoons, compared to only 68% for the BNR system. The resulting increase in global warming potential and cumulative energy demand for the RABR system were only 26% and 42%, respectively, compared to 174% and

186% for the BNR system. This study demonstrates the merit of the novel RABR systems when land intensive wastewater treatment strategies are acceptable.

Introduction

In an effort to protect fresh water resources, the wastewater industry is working to reduce nutrient discharges from treatment plants to surface waters across the United States. The most reliable and cost effective method of nutrient removal is generally achieved through biological processes termed biological nutrient removal (BNR) activated sludge. These systems utilize mixed consortia of bacteria and by control of the biochemical environment (i.e. aerobic, anaerobic, or anoxic) can achieve both nitrogen and phosphorus removal. While the intent is environmental protection, life cycle assessment (LCA) studies have acknowledged that the increased electricity requirements from aeration and pumping in these systems can shift the environmental burden from local water quality impacts to global warming impacts (Corominas et al., 2013; Foley et al., 2010; Godin et al., 2012). To offset these impacts, many plants also employ anaerobic digestion of waste biosolids and collect biogas to generate heat and/or electricity on-site.

Recent interest in algae for energy production and other biomanufacturing applications has led to an increased push to utilize nutrients in wastewater, rather than fertilizers, to produce algal biomass thus lessening the life cycle impacts of these systems (Christenson and Sims, 2011; Clarens et al., 2010; Pittman et al., 2011). This provides an opportunity to use algaculture, rather than BNR, for nutrient removal. However, the land requirements for algaculture are much larger than that of activated sludge systems; therefore algal nutrient removal may not be feasible in urban areas where land availability is limited. Alternatively, algaculture may provide an exciting opportunity for energy-efficient nutrient removal for wastewater systems in rural areas with available land. Steele et al. (2014) showed that for small activated sludge systems without nutrient removal, algae can offset the environmental burdens of wastewater treatment more effectively than upgrading the system using more conventional nutrient removal strategies.

Algaculture may be particularly appealing for communities that already have lagoon systems in place. Lagoons are a simple treatment approach that operate based on natural processes and are attractive to small, rural communities because of their reliability and ease-of-operation, but removal of ammonia and phosphorus is unreliable (EPA, 2002) thus new systems must be considered to meet nutrient discharge limits. Rotating algal bioreactors (RABRs) are a novel type of attached growth algaculture that use an algal biofilm with similar configuration to rotating biological contactors (RBCs) used in wastewater treatment with bacterial biofilms; the cylindrical base with cotton growth substratum (Figure 4.1) is partially submerged in wastewater and the rotating action alternately exposes the biofilm to wastewater nutrients and air. This approach helps expose the algal biomass to sunlight without limiting the depth of the reactor, unlike other algal wastewater treatment technologies (e.g. raceway ponds). Preliminary studies have shown that RABRs have the potential to decrease nutrient concentrations in wastewater while generating an algal biomass product that is easily harvested from the cotton substratum and can be used for the production of valuable bioproducts (Christenson and Sims, 2012). The ease of harvesting, a process which is typically problematic for algaculture systems (Uduman et al., 2010), is an important benefit of using the RABR design because it produces a concentrated algal slurry with minimal energy inputs.

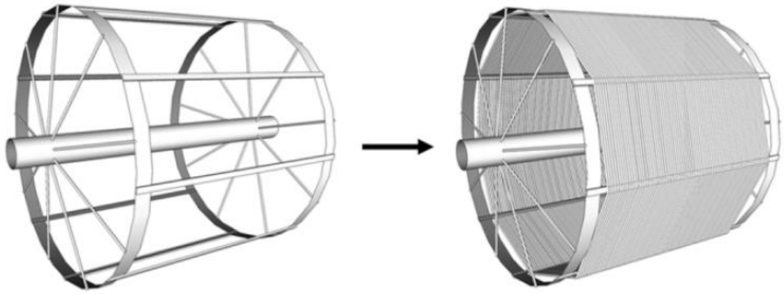


Figure 4.1: RABR aluminum frame (left) and with cotton substratum (right) (Christenson and Sims, 2012).

One example of a location with tightening nutrient discharge limits is Utah, where the Utah Department of Environmental Quality (UDEQ) wants to limit the amount of nitrogen and phosphorus that is discharged from publicly owned treatment works (POTWs) to surface waters across the state. Of the 57 POTWs in Utah, 27 are lagoon systems which are mostly small ($\leq$ 2 million gallons per day [MGD]) with the exception of the Logan lagoons, which are rated to treat up to 19 MGD. Studies were performed to determine the life cycle costs of tightening nitrogen and phosphorus limits; it was estimated that upgrading these systems would cost between $113 million and $166 million for Logan and between $8 and $13 million (net present value in 2009 dollars) for each small system, depending on the stringency of the new nitrogen and phosphorus limits (CH2MHill, 2010a, 2010b). Additionally, a cost estimate comparison was performed for preliminary designs of a combined BNR/chemical nutrient removal system and an algal biofilm reactor solution for the Logan lagoons (Carollo Engineers, 2013). It was predicted that the algae-based system would have higher construction and lower operations and maintenance (O&M) costs than the BNR system ($239 versus $111 million for construction; $4 versus $5 million for O&M), although this analysis did not consider the potential revenue from algal biomass nor had the design of the RABR been optimized for scaling economically. These systems, if implemented, serve to protect water quality, but impart high capital and operations and maintenance costs to local and state governments. In addition to cost considerations, it is critical that environmental concerns also be taken into account when implementing new wastewater treatment strategies, both in Utah and elsewhere, and this study aimed to investigate these impacts from a life cycle perspective.

Goal and Scope Definition

The goal of this study was to compare life cycle environmental impacts associated with upgrading existing lagoon systems using two alternative treatment processes designed to comply with stringent pollutant discharge standards, including limits on phosphorus and nitrogen, which are becoming common across the United States. Two systems were compared: (1) an upgrade of RABRs to an existing lagoon system (the L-RABR scenario) and (2) construction of a BNR activated sludge system (the BNR-AS scenario). The purpose was to supply information for wastewater engineers and regulatory bodies to determine whether installing RABRs at existing lagoon treatment plants is a viable option for improving effluent water quality while minimizing other environmental impacts and developing an algae-based biomanufacturing industry when compared to construction of new activated sludge plants. The system was modeled after the Logan lagoons and the design of a BNR plant proposed to replace the lagoons (Figure 4.2) (Carollo Engineers, 2013).The functional unit was defined as the treatment of influent raw wastewater flow over 20 years, as described in Foley et al. (2010). Volumetric flow (Figure 4.3) is based on four years (2010-2013) of daily flow at the Logan lagoons [see Appendix C1] and influent quality parameters (Table 4.) are based on values proposed in the Logan Wastewater Master Plan (Carollo Engineers, 2013). Study systems are designed to meet discharge limits on ammonia and phosphorus (Table 4.2) proposed for Logan. Because phosphorus is regulated on the basis of cumulative seasonal mass discharge rather than concentration, the daily mass discharge limit was determined and target effluent concentration was calculated using influent volumetric flow (see Appendix C2).

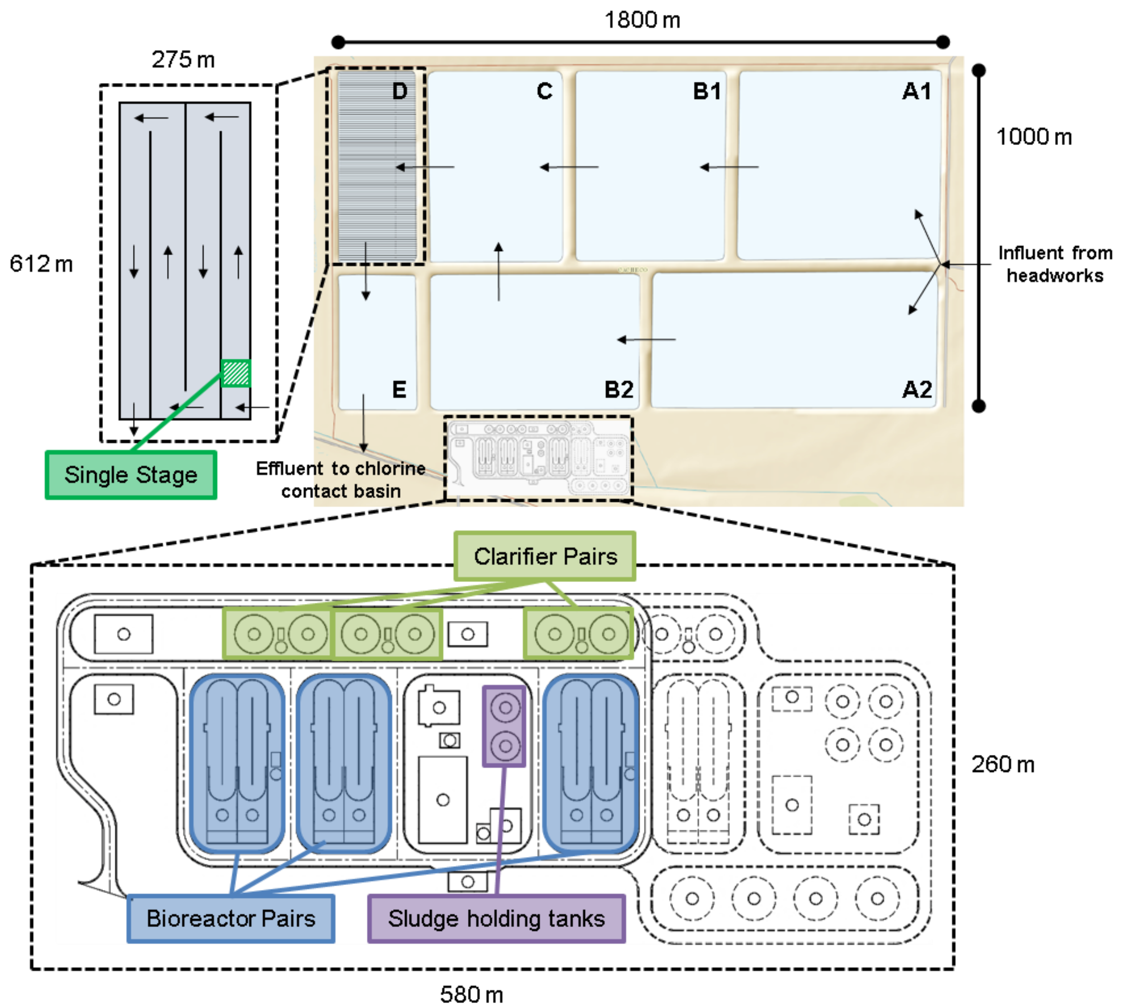


Figure 4.2: Plan view of existing lagoon showing location and size of L-RABR (top left) and BNR-AS (bottom) upgrade scenarios. Note the differences in scale between two scenario insets. Dashed lines within the BNR-AS scenario diagram represent long-term potential upgrades (primary clarifiers, additional bioreactor and clarifier pairs, and anaerobic digestion tanks) proposed in (Carollo Engineers, 2013) but not modeled in this study.

The system boundaries were drawn to include wastewater entering the facility and all discharges to the receiving environments, as well as construction of the upgraded systems

(Figure 4.4). Construction impacts considered only raw materials needed to build each scenario, as the process of constructing the L-RABR system is not yet well defined. Impacts from operation include electricity and alum used for treatment, discharge of treated effluent, and disposal of waste biosolids. Disinfection by chlorination and landfilling of residuals from headworks were not considered, as they were assumed to be equal between the two scenarios. It is suggested, however, that the effects of algal WWT on disinfection be tested experimentally and considered in future analysis because the high pH caused by photosynthetic activity can made chlorine less effective and may cause an increased use of chlorine or necessitate pH adjustment prior to chlorination. Treated effluent is discharged into wetlands, which flow to the Cutler Reservoir. Waste biosolids are landfilled. First order (i.e. direct) and second order (i.e. from upstream and downstream processes) emissions are considered for the construction and operation stages of the POTW. Life of the plant was modeled to be 20 years, however, end-of-life of the systems was excluded due to the relatively small impacts compared to construction and operation phases (Emmerson et al., 1995; Zhang and Wilson, 2000) which is consistent with other wastewater LCAs (Foley et al., 2010). The end use of algal biomass from the RABR system was also excluded to retain the focus on wastewater treatment processes, but potential biomanufacturing and bioprocessing applications were included in the discussion.

Table 4.1: Design influent quality parameters

	Winter	Summer
Biochemical Oxygen Demand (BOD) (mg/L)	140	100
Total Suspended Solids (TSS) (mg/L)	180	113
Ammonia (mg NH_3/L)	22	17
Phosphorus (mg P/L)	6.3	4.0
Temperature (°C)	13	18

**Winter: Nov-Apr, Summer: May-Oct

Table 4.2: Design discharge limits

	Season*	Winter	Spring	Summer	Fall
Ammonia	Monthly Ave (mg/L)	3	3	1.3	2.6
	Daily Max (mg/L)	5	8	6	7
	Season**	**Winter**		**Summer**	
Phosphorus	Total Discharge (kg/season)	12,901		11,487	
	Daily Discharge (kg/day)	71.3		62.4	

*Winter: Dec-Feb, Spring: Mar-May, Summer: Jun-Aug, Fall: Sep-Nov
**Winter: Nov-Apr, Summer: May-Oct

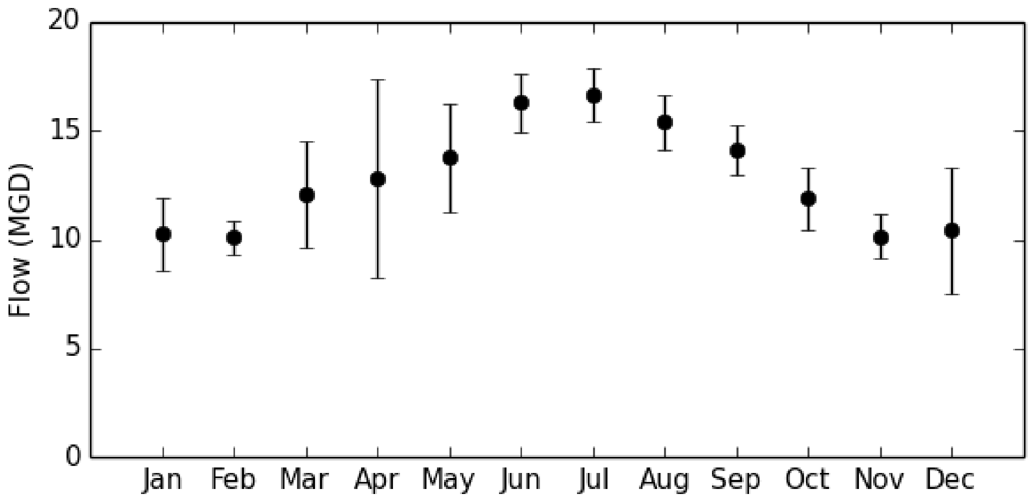


Figure 4.3: Monthly volumetric flow data (2010-2013). Error bars represent one standard deviation above and below mean.

The impact assessment method used was TRACI 2.1 (Bare, 2012) and was performed using Excel (Microsoft, 2013), GaBi 6.4.1.20 (PE International), and Python (Python Software Foundation, v 2.7). Eutrophication potential (EUT) in freshwater, global warming potential (GWP), and cumulative energy demand (CED) were the impact categories considered. Data for construction stage materials and operation stage alum and electricity use were obtained from Ecoinvent Integrated v2.2, accessed within GaBi. Electricity data were modeled as average of US electricity at grid. Methane emissions from landfilling sludge were calculated according to the EPA's Greenhouse Gases (GHG) Reporting Program (40 CFR part 98, subpart HH). More information is given in Appendix C8.

Modeling and Design Approach

Lagoon-Rotating Algal Biofilm Reactor System

The L-RABR system was designed around the existing lagoons currently used for the treatment of municipal wastewater in Logan, Utah. The lagoons (Figure 4.2) are divided into seven ponds; flow travels from the A ponds to the B ponds in parallel, then combines in pond C before flowing into ponds D and E. Surface aeration is used in ponds A1 and A2. In the design of the L-RABR system, RABR units were installed in pond D to achieve nitrogen and phosphorus removal. In the pond, 612 meters (2,008 feet) long by 275 meters (903 feet) wide, three 600 meter walls were built length-wise to achieve plug-flow conditions, creating four channels that were 65 meters wide by 612 meters long, a total channel length of 2,448 meters. The channels were then further divided into stages using baffles across the width of the channel, as is common with RBCs used for wastewater treatment (Grady et al., 2011). Stages consist of 875 individual RABR units covering an area of 3,900 square meters (65 meters wide by 60 meters long) each; thirty-five RABR units are mounted across the width of each channel perpendicular to flow, and twenty-five rows, spaced 2.4 meters (8 feet) apart, are included per stage. The number of stages necessary to achieve effluent quality goals varied each month with influent water quality changes.

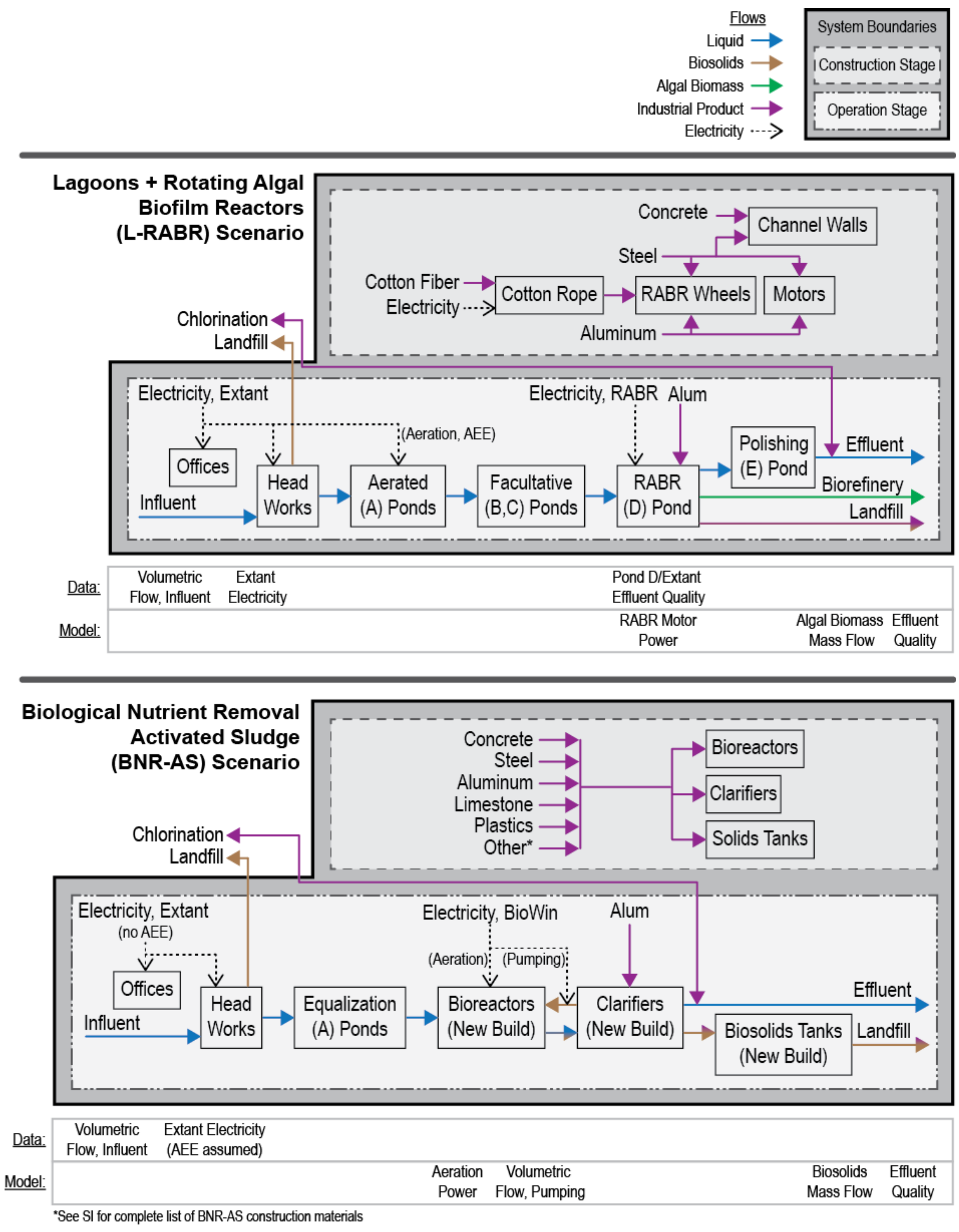


Figure 4.4: System diagrams for L-RABR (top) and BNR-AS (bottom) scenarios showing what information came from historical data and models. Flows outside of the system boundaries were identical between both scenarios and thus excluded from analysis.

Nitrogen and phosphorus removal by the L-RABR system was modeled in Python, using concentration data collected weekly from 2010 to 2013 at the lagoons as growth media conditions. Nutrient removal kinetics were determined from a combination of lab- and pilot-scale studies performed at USU (see Appendix C3). Nitrogen removal was shown to be concentration dependent, and thus was modeled using pseudo first-order kinetics and a removal rate of 0.461 d^{-1}. Phosphorus removal was not concentration dependent, thus was modeled as a zero-order reaction and a removal rate of 0.379 mg L^{-1} d^{-1}. Each stage was modeled as a continuous stirred-tank reactor with simultaneous N and P removal, supported by the assumption that channels installed in pond D and baffles installed downstream of each stage would mimic these conditions. The effect of temperature on algal growth and nutrient removal was not included in the RABR model, however it is suggested that local climate considerations be addressed in the future. Alum was used to precipitate phosphorus in a polishing step, and alum use was calculated using a 2:1 molar ratio for aluminum to phosphorus (Tchobanoglous et al., 2003).

Direct methane emissions were estimated based on predicted COD removal in the lagoons (M. R. J. Doorn et al., 2006). Energy use by the RABRs was calculated based on 6 watt motors used to turn each RABR unit (Christenson and Sims, 2012). The total energy use for the L-RABR scenario was the sum of the extant energy use reported for the Logan lagoons and the energy use by the RABRs. Materials for construction were estimated based on pilot-scale unit supplies. More information on L-RABR design, treatment modeling, energy use calculations, and construction materials is provided in Appendix C3.

Biological Nutrient Removal Activated Sludge System

The BNR-AS system is based on the conceptual design proposed in the City of Logan's wastewater master plan (Carollo Engineers, 2013). The system utilizes three stage (anaerobic-anoxic-aerobic) biological nutrient removal bioreactors (Figure 4.5) followed by sedimentation. The plant consists of six bioreactors in parallel with the total anaerobic, anoxic, and aerobic tank volumes equaling 1.0, 3.0, and 8.9 million gallons, respectively; six secondary clarifiers, 80 feet in diameter; and two sludge holding tanks, 60 feet in diameter. Waste activated sludge is stored in the sludge holding tanks before being landfilled. The existing lagoons are used for equalization. As with the L-RABR system, alum is used to chemically precipitate phosphorus when limits are not able to be met with the biological treatment system.

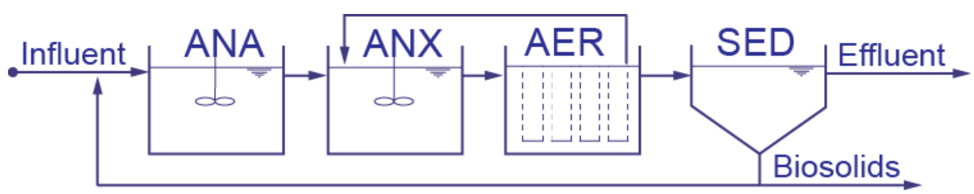


Figure 4.5: Process flow schematic for BNR-AS scenario, showing one of six treatment trains modelled in parallel.

The system was modeled using BioWin 4.0 (Envirosim) to determine effluent quality, waste sludge production, pumping requirements, methane and nitrous oxide emissions and power required for aeration. The target solids retention time is 20 days. Mixed liquor recycle rate is four times the influent flow rate. Aeration is provided by surface aerators with a standard oxygen transfer rate of 3.5 lbs O_2/hp-hr.

Energy use by the activated sludge system was estimated from pumping and aeration requirements reported by BioWin; other energy use (for offices and headworks) was estimated from extant energy reported for the Logan lagoons excluding contribution of surface aeration in the ponds (AEE, assumed to be 25% of the monthly energy use). Pumping requirements were based on flow rates for returning activated sludge from secondary clarifiers to bioreactors. Power required for surface aerators were also included. Construction materials were estimated using the procedure outlined previously (Foley et al., 2010). Concrete required for plant structures was determined from engineering drawings, then other materials were estimated using multipliers (per volume of concrete) determined in a comprehensive life cycle inventory of wastewater treatment plants (Doka., 2003). More information on BNR-AS design, treatment modeling, energy use calculations, and construction materials is provided in the Appendix C4.

Impact Assessment Methods

Eutrophication potential (EUT), global warming potential (GWP), and cumulative energy demand (CED) were calculated for the extant lagoon and both upgrade scenarios (L-RABR and BNR-AS). Impact assessment was performed using a net impact approach, based on the net environmental benefit approach (NEB) (Godin et al., 2012; Steele et al., 2014). Briefly, this approach considers the impacts from a no action scenario (i.e. continued use of the extant lagoons) and subtracts those impacts from the realized impacts from treated wastewater and plant operation to determine the benefit of the processes considered. The net impact is the sum off impacts from all stages minus the benefit.

Operation Stage Impact Assessment

The EUT category considered effects from effluent discharged COD, TN, and TP, as well impacts from alum production and electricity generation. The GWP category considers GHGs from direct, secondary, and background emissions. Direct emissions for the extant lagoon and L-RABR scenario include methane from the lagoons due to the anaerobic zones within the lagoons. Methane (reported in BioWin) and nitrous oxide (a by-product of denitrification) are the direct emissions considered from the BNR-AS scenario. Secondary emissions from COD and TN in discharged in effluent were considered for all scenarios and methane from landfilling sludge was considered for the upgrade scenarios. Background emissions from production of alum and electricity generation were considered when applicable. The CED category considers primary energy demand required for alum production and electricity generation. More information can be found in the Appendix C8.

Construction Stage Impact Assessment

No transportation or earthwork, only materials, were considered when generating the inventory for the construction phase. The construction of the L-RABR scenario considers installation of channel walls in pond D, RABR wheel frames and shafts, motors, and cotton substratum to support biofilm. The volume of concrete was estimated for the channel walls and used to determine reinforcing steel as previously described (Foley et al., 2010). Other materials for RABRs were calculated by scaling up the pilot system used at USU. Construction materials for the BNR-AS scenario were determined based on the volume of concrete necessary to construct the system as described (Foley et al., 2010). Additionally, the RABR pond may need to be covered in greenhouse-like material for heat retention to prevent freezing of the wheels during winter months in some locations. Though it was not considered in this study, it should be addressed when and L-RABR system is considered for certain climates.

Variability/Uncertainty

Monte Carlo analysis (MCA) was used to address uncertainty in the influent characteristics of the wastewater to each scenario. One thousand simulations were run for each month to allow extrapolation of impact assessment results, on a yearly basis, over the lifetime of the plant. The distributions for each influent parameter used in generating the conditions for MCA were determined from the distributions of data from the Logan plant.

Volumetric flow data were used in the treatment model as an estimate of the influent flow for both L-RABR and BNR-AS scenarios. The gamma distribution for each month was determined by estimating shape and rate parameters (alpha and beta, respectively) based on reported data from each month; the resulting probability density function of each gamma distribution (Figure C6) was then used to generate 1000 values of influent flow for each month during MCA.

Effluent quality data from the existing lagoon system were used to determine the influent characteristics for the RABR treatment model because no changes were made to the lagoons in the L-RABR scenario. Due to a lower sampling frequency for water quality parameters than for volumetric flow in the provided data, there was not enough data to determine the expected distribution for each month. Therefore, all of the data for each of the parameters BOD, TSS, and TP were used to estimate alpha and beta for each parameter's gamma distributions; the probability density function of each gamma distribution (Figures C7-C9) was then scaled to the range observed for each month and used to generate 1000 values of influent flow for each month during MCA. Ammonia showed a stronger seasonal effect than other quality parameters, with significantly different values and distributions for summer (May – October) and winter (November – April). Therefore, normal distributions (Figure C10) were used for ammonia concentrations during MCA.

Influent quality characteristics for the BNR-AS scenario were taken from design values and thus the distribution of these parameters was unknown. Therefore BOD, TSS, and NH_3 concentrations were modeled as normal distributions, using the distribution of these parameters in the effluent data to estimate the standard deviations necessary. However, since effluent phosphorus is often a function of the influent P concentration, particularly for lagoon systems, the influent TP values were generated using the same gamma distribution created from effluent data and scaled to reflect estimated influent concentrations. AutoIT (v3.3.14.2) was used to automate the 1000 runs in BioWin. More information in Appendix C7.

Inventory Results

The treatment models were used to estimate the effluent quality, chemical and electricity use, algal biomass production, and biosolids to landfill for the L-RABR and BNR-AS scenarios. The results and uncertainty are represented in box and whisker plots (Figure 4.6 - Figure 4.11). In some cases, the results from design values did not match the median values for MCA results; for these figures, solid black circle and diamond markers represent model results for design values of L-RABR and BNR-AS scenarios, respectively. Uncertainty, based on 1000 runs of Monte Carlo analysis, is shown as box and whisker plots; boxes represent the interquartile range ($IQR = Q_3 – Q_1$) and show the median (Q_2) result; whiskers extend to the furthest data point within one IQR above or below Q_3 and Q_2, respectively.

Effluent Qualtiy

Both the L-RABR and BNR-AS scenarios were able to achieve nutrient removal to some extent. Total nitrogen (TN) results show that L-RABR outperformed BNR-AS (Figure 4.6). However, BNR-AS was more consistent in removing ammonia to below discharge permit limits, with very little variation (Figure 4.7), but both scenarios were able to reach permit limits in most cases.

There were limitations in the RABR treatment model in terms of TN, as data for lagoon effluent and lab- and pilot-scale RABR tests mainly evaluated ammonia rather than other nitrogen species. No effluent nitrate was considered from the extant lagoon or L-RABR scenario because nitrification is not a dominant biological conversion in lagoon systems (Middlebrooks et al., 1999). Additionally, three pilot-scale tests did include data for nitrate and nitrite, and two of these three tests showed promising results with effluent nitrate concentrations of 3.0 and 1.1 mg/L and both nitrite concentrations were <0.1 mg/L. A third test had higher effluent nitrate and nitrite (24.5 and 0.5 mg/L respectively), but was run with a less developed algal biofilm. Therefore, further tests are required to confirm the ability of RABRs to achieve low TN. It is also recommended that algal biomass harvesting be performed on a rotating schedule to ensure a consistently mature biofilm is maintained in each RABR stage.

When not considering chemical precipitation (alum addition) phosphorus removal was better in the L-RABR scenario than with activated sludge (Figure 4.8), meeting discharge limits in >50% of cases in most months (Figure 4.9). The design proposed for BNR-AS (Carollo Engineers, 2013) included chemical precipitation, so alum use was calculated to achieve necessary P removal to meet discharge limits (Figure 4.10). Alum use was also calculated for L-RABR scenario when necessary. The limited P removal by the BNR-AS system could also be addressed through improvements to the proposed design. Notably, the low BOD:P ratio of the influent wastewater causes limitation of biological phosphorus accumulation; thus providing the system with a source of volatile fatty acids (by fermenting solids or from an external source) could greatly improve the BNR-AS system's performance in terms of P removal.

Alum and Electricity Use

Alum use was calculated for both scenarios using a 2:1 molar Al:P ratio (Figure 4.10). Alum was necessary for all months in the BNR-AS scenario, but not in most winter months for the L-RABR scenario. In May, August, September, October, and January alum was necessary with some influent conditions modeled, but was not necessary at design influent conditions. In June and July, alum use between the two scenarios was similar.

Electricity use was calculated for running each treatment scenario (Figure 4.11). L-RABR electricity was calculated based on 6 W per RABR unit, as described previously (Christenson and Sims, 2012). Motors to run RABR wheels only marginally increased electricity use from the extant lagoons. Electricity use by the BNR-AS was much higher than the extant lagoons and the L-RABR scenario.

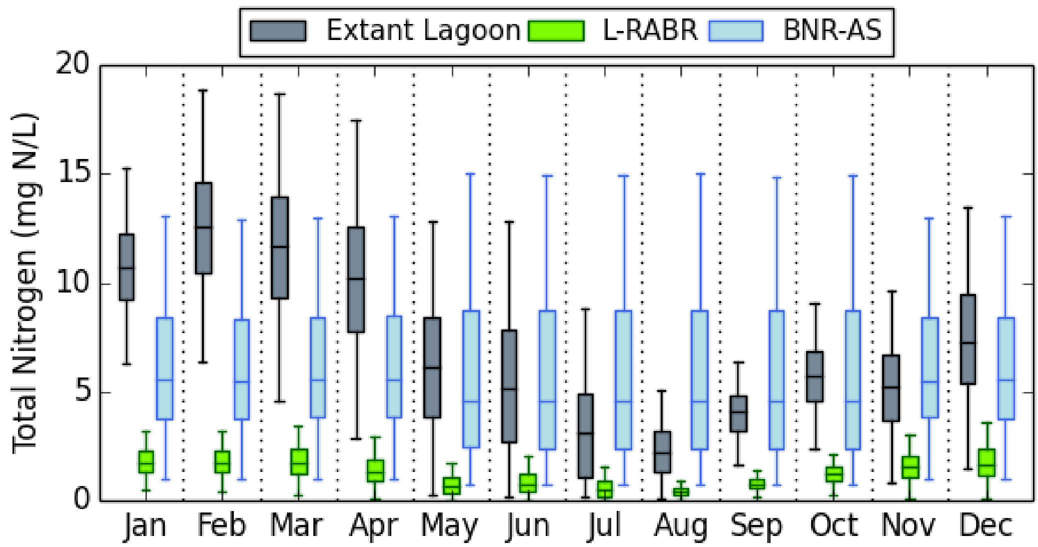


Figure 4.6: Effluent total nitrogen (TN) concentrations showing distributions for the extant lagoon (grey), L-RABR (green), and BNR-AS (blue) scenarios. Design influent TN was 19 and 24 mg/L in summer and winter respectively.

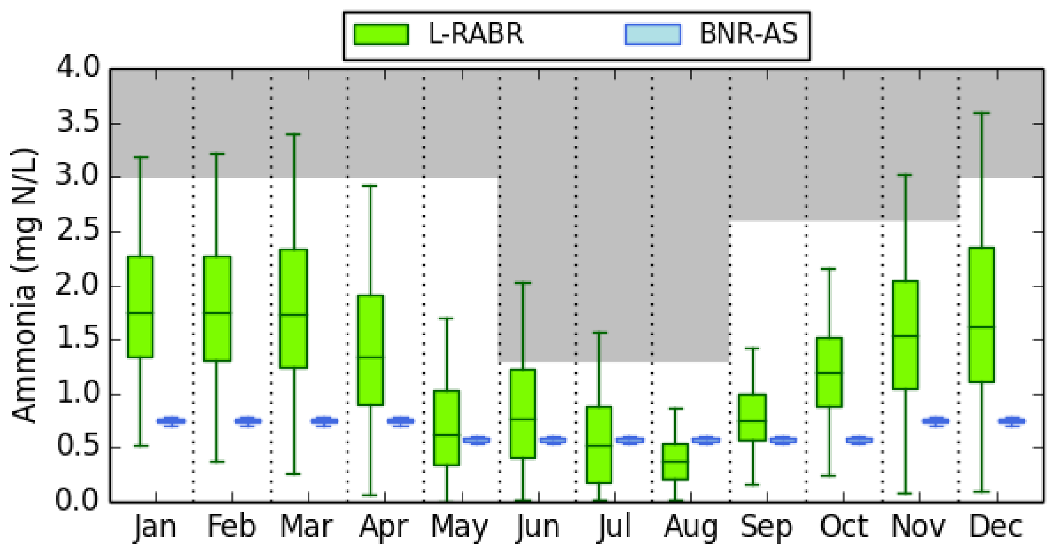


Figure 4.7: Effluent ammonia for L-RABR and BNR-AS scenarios. Shaded area represents values which exceed discharge limits.

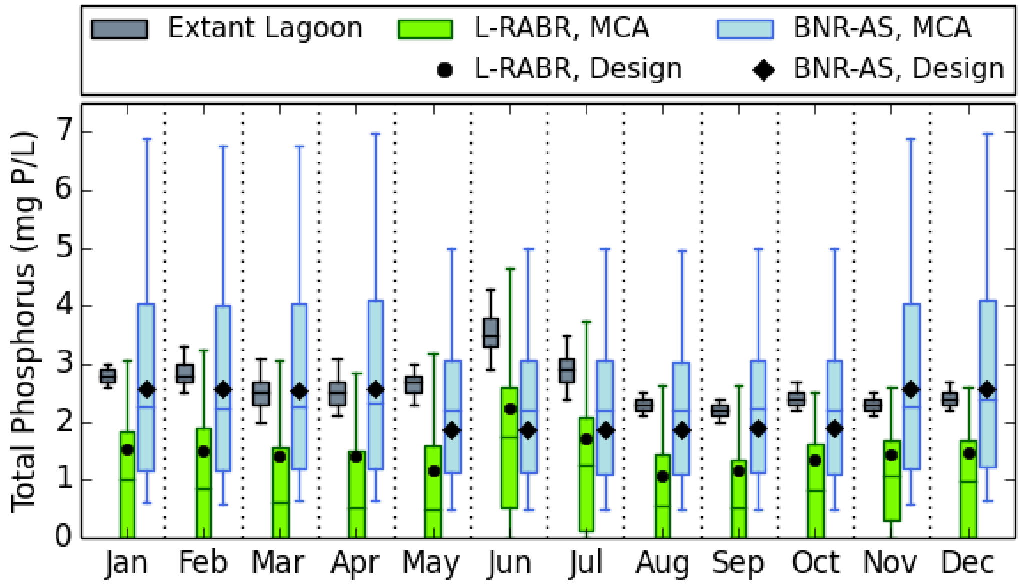


Figure 4.8: Total phosphorus concentrations (mg P/L) before precipitation with alum. Distributions for the extant lagoon (dark grey), L-RABR (green), and BNR-AS (blue) scenarios are shown. Single-point, design-value model results are also shown for L-RABR (circle) and BNR-AS (diamond) scenarios. Design influent TP was 4.0 and 6.3 mg/L in summer and winter respectively.

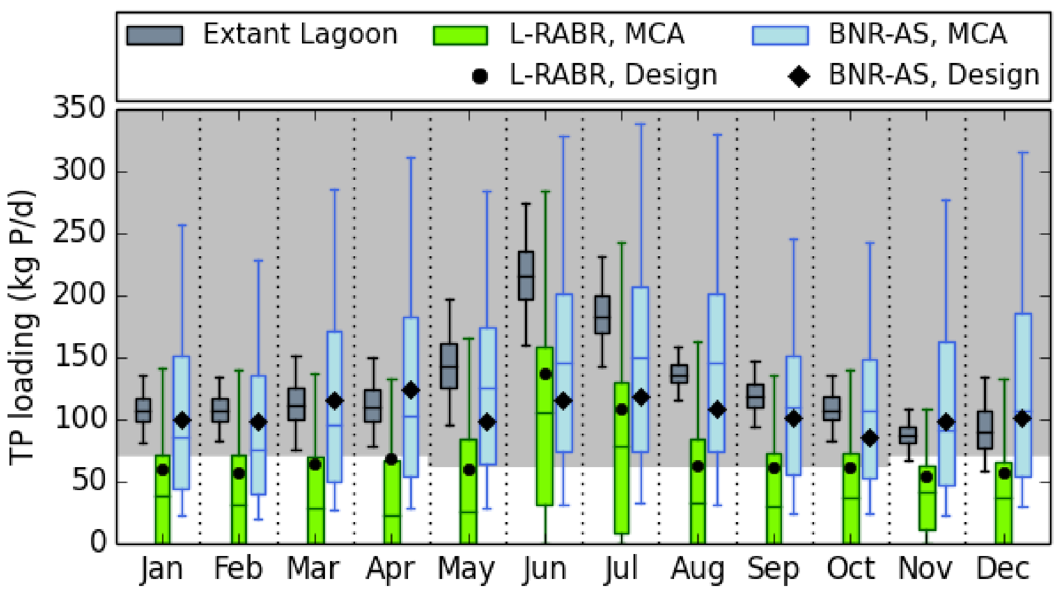


Figure 4.9: Phosphorus loading (kg P/d) without alum use showing distribution for the extant lagoon (dark grey), L-RABR (green), and BNR-AS (blue) scenarios. Single-point, design-value model results are also shown for L-RABR (circle) and BNR-AS (diamond) scenarios. Shaded area represents values which exceed discharge limits.

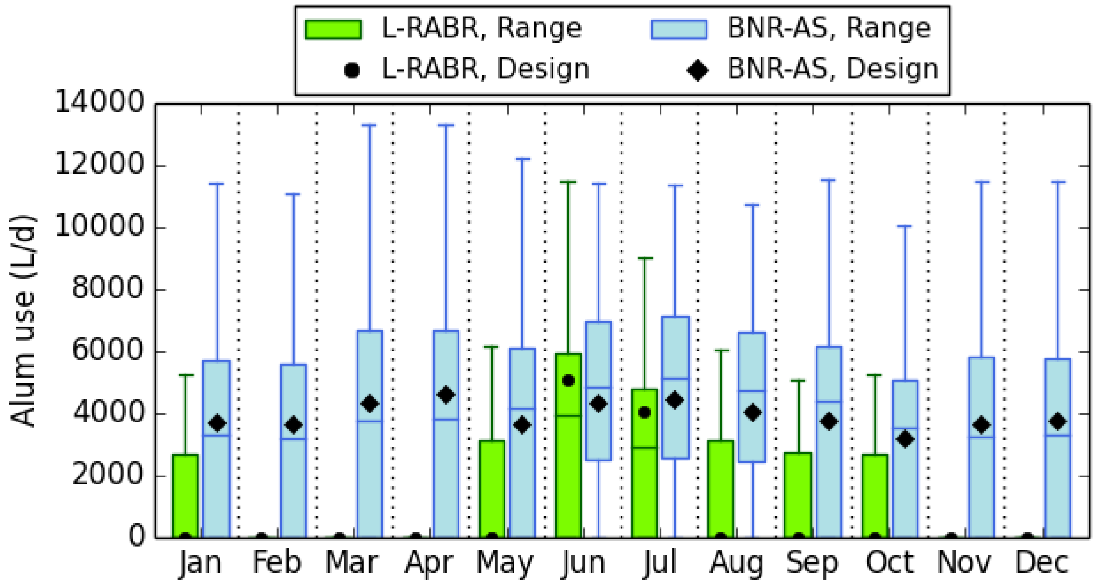


Figure 4.10: Alum use for L-RABR and BNR-AS scenarios, showing distributions (green and blue) and single-point, design-value model results (circles and diamonds).

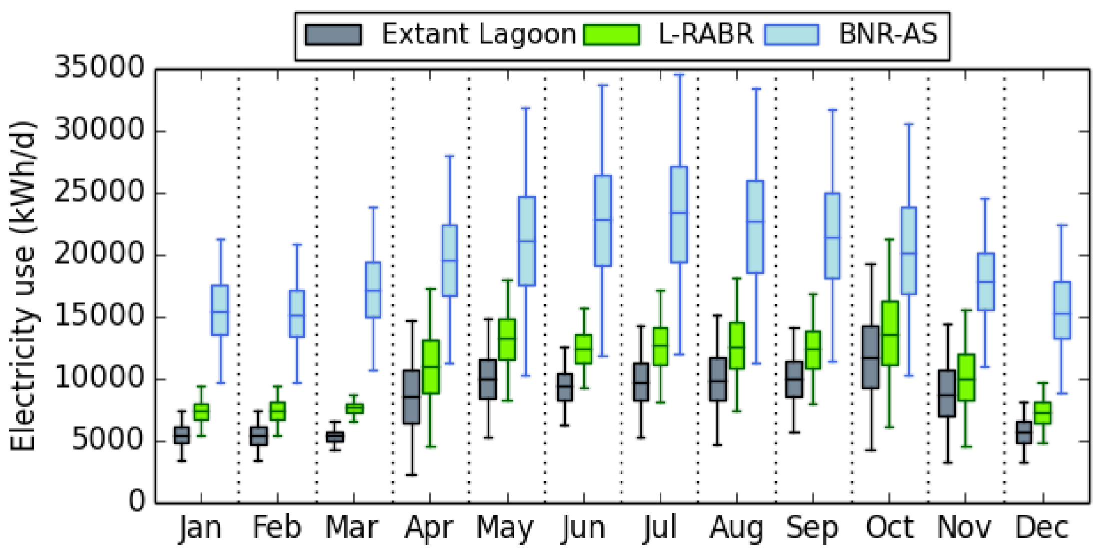


Figure 4.11: Electricity use for BNR-AS scenario (blue) and L-RABR scenario (green).

<u>Impact Assessment</u>

Results from the impact assessment suggest that the L-RABR scenario causes fewer impacts than the BNR-AS system as an upgrade to existing lagoons in all categories considered in this study (Table 4.3). The EUT, GWP, and CED impacts for construction of the L-RABR scenario were 83, 73, and 61% lower than for the BNR-AS scenario; operation phase impacts were 53, 54, and 50% lower for L-RABR. Additionally, the avoided EUT and GWP impacts (or benefits) seen by the L-RABR scenario were 20% and 108% greater than for the BNR-AS scenario. These results represent a conservative analysis with respect to the potential beneficial uses of by-products from treatment. In the L-RABR scenario, algal biomass could potentially be used as a feedstock for renewable energy or to replace other fossil-carbon based products such as plastics or industrial solvents. Biogas utilization could also be used to reduce the impacts of the BNR-AS scenario, either through on-site anaerobic digestion or landfill gas capture and utilization.

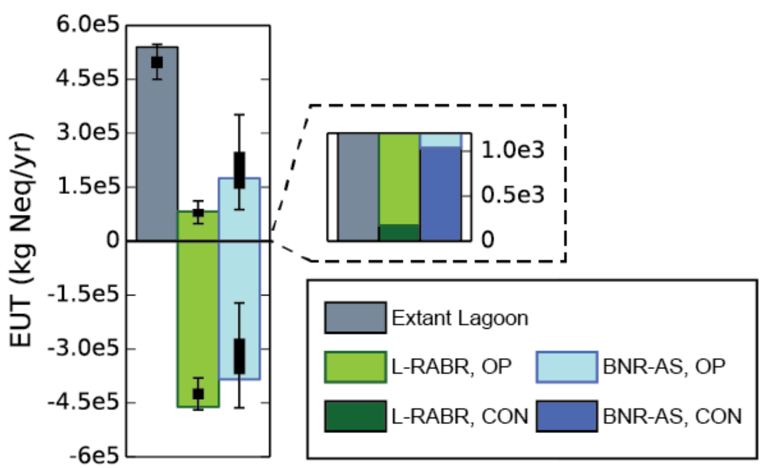


Figure 4.12: Eutrophication potential showing avoided (-) and realized (+) impacts for operation (OP) and construction (CON) stages for all scenarios. Colored bars represent design-value results; error bars show IQR results (thick bars) and furthest data point within one IQR above and below IQR (whiskers).

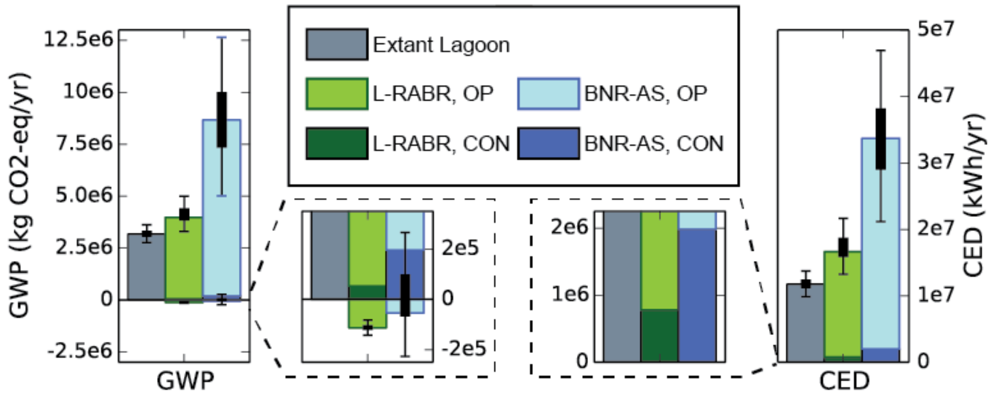


Figure 4.13: Global warming potential (left) and cumulative energy demand (right) showing avoided (-) and realized (+) impacts for operation (OP) and construction (CON) stages for all scenarios. Colored bars represent design-value results; error bars show IQR results (thick bars) and furthest data point within one IQR above and below IQR (whiskers).

Table 4.3: Impact assessment results showing benefits (-), construction impacts (+), operation impacts (+), and net impact for all scenarios and categories considered.

		Extant Lagoon	L-RABR	BNR-AS
Benefit	**EUT (kg N-eq/yr)**	-	-4.61E+05	-3.84E+05
	GWP (kg CO_2-eq/yr)	-	-1.13E+05	-5.43E+04
Construction	**EUT (kg N-eq/yr)**	-	1.76E+02	1.04E+03
	GWP (kg CO_2-eq/yr)	-	5.31E+04	1.95E+05
	CED (kWh/yr)	-	7.81E+05	1.98E+06
Operation	**EUT (kg N-eq/yr)**	5.39E+05	8.18E+04	1.73E+05
	GWP (kg CO_2-eq/yr)	3.17E+06	3.93E+06	8.47E+06
	CED (kWh/yr)	1.18E+07	1.59E+07	3.17E+07
Net Impact	**EUT (kg N-eq/yr)**	5.39E+05	-3.79E+05	-2.10E+05
	GWP (kg CO_2-eq/yr)	3.17E+06	3.87E+06	8.61E+06
	CED (kWh/yr)	1.18E+07	1.66E+07	3.37E+07

The larger EUT benefits and smaller EUT impacts for the L-RABR system (Figures 4.12 and 4.13) are due largely to the fact that the BNR-AS scenario achieves nitrification but not complete denitrification, thus effluent nitrate contributes to eutrophication as both a realized impact (Figure 4.14) and by reducing the extent of the benefit in this category. The large difference in construction impacts between the scenarios is partially a consequence of a more thorough inventory of materials used in the BNR-AS scenario, particularly for copper which contributes only a small portion of the mass in the BNR-AS system but a large fraction of the EUT impact (Figure 4.15) and was not considered in the L-RABR but would likely be necessary in small quantities for wiring the RABRs.

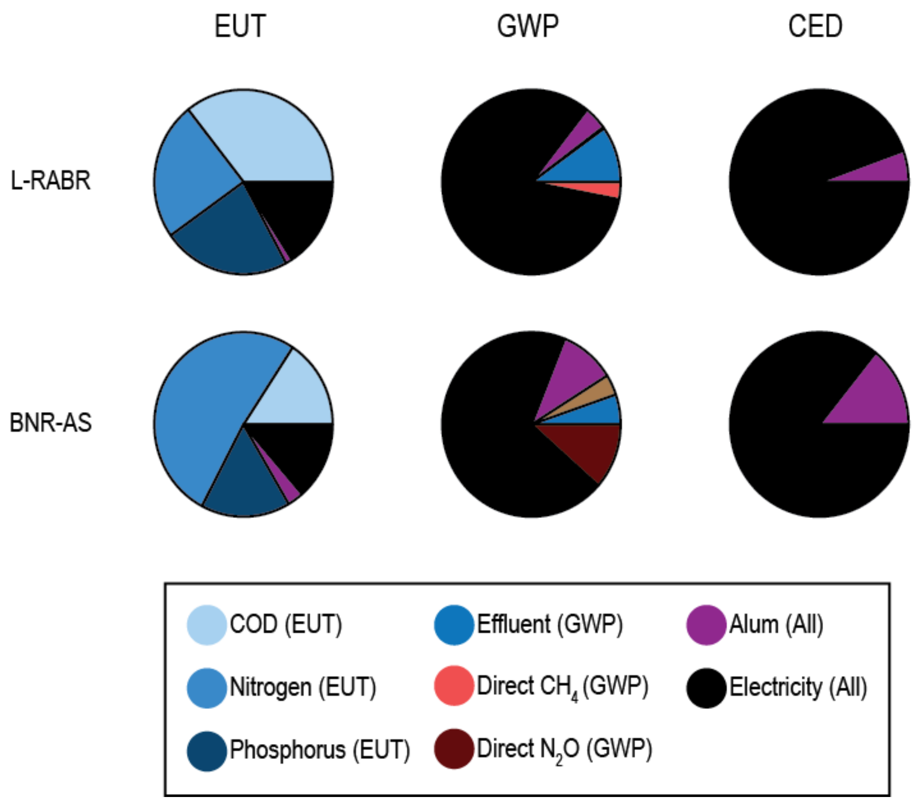


Figure 4.14: Contributions of processes and flows to operation stage impacts for both upgrade scenarios and all impact categories. See Table 4.3 for impact values.

Operation Stage Impacts

Contributions of processes and flow to each impact category were considered for the operation phase. Comparing eutrophication impacts, both realized and avoided, confirms the benefit of upgrading the lagoon system to include nutrient removal (Figure 4.12). Both L-RABR and BNR-AS scenarios show strongly that these systems provide a net benefit to the environment, which is the purpose of wastewater treatment. Conversely, the impacts of electricity and alum use in both GWP and CED categories (Figure 4.14) highlights the trade-offs that are necessary to consider for implementation of advanced treatment systems.

Construction Stage Impacts

Construction stage impacts accounted for only a small fraction of the gross impacts in each category for both scenarios contributing only 0.2, 1.3, and 4.7% of the gross EUT, GWP, and CED impacts in the L-RABR scenario and 0.6, 2.3, and 5.9% in the BNR-AS scenario. These values represent an underestimate of the actual impacts that would occur if either scenario was implemented because they only consider production of the materials used in construction, not transportation to the site, earthwork on site, or other construction processes. Nevertheless, the impacts for these excluded processes would largely be a function of the mass of materials used in construction and thus these results provide enough information to compare the two scenarios and extrapolate the findings to other potential lagoon systems considering nutrient-removal upgrades.

Although small relative to impacts from the operation stage, it is important to consider the impacts of construction materials to the overall life cycle impacts of systems, especially for technologies in the early stage of development, such as RABRs. Based on the contribution of the materials used in RABRs, impacts from construction materials can be minimized by maximizing the amount of recycled aluminum used in building the RABR frames (Figure 4.15). Although cotton did not contribute significantly in any of the impact categories considered in this study, a more thorough investigation of the impacts of this cotton use are suggested as impacts from cotton are highly dependent on the properties of the material (Van Der Velden et al., 2014) and it likely has high impacts in categories such as water and land use that are not necessarily relevant to wastewater treatment plants but would become more relevant if RABR systems were to be implemented on a larger scale.

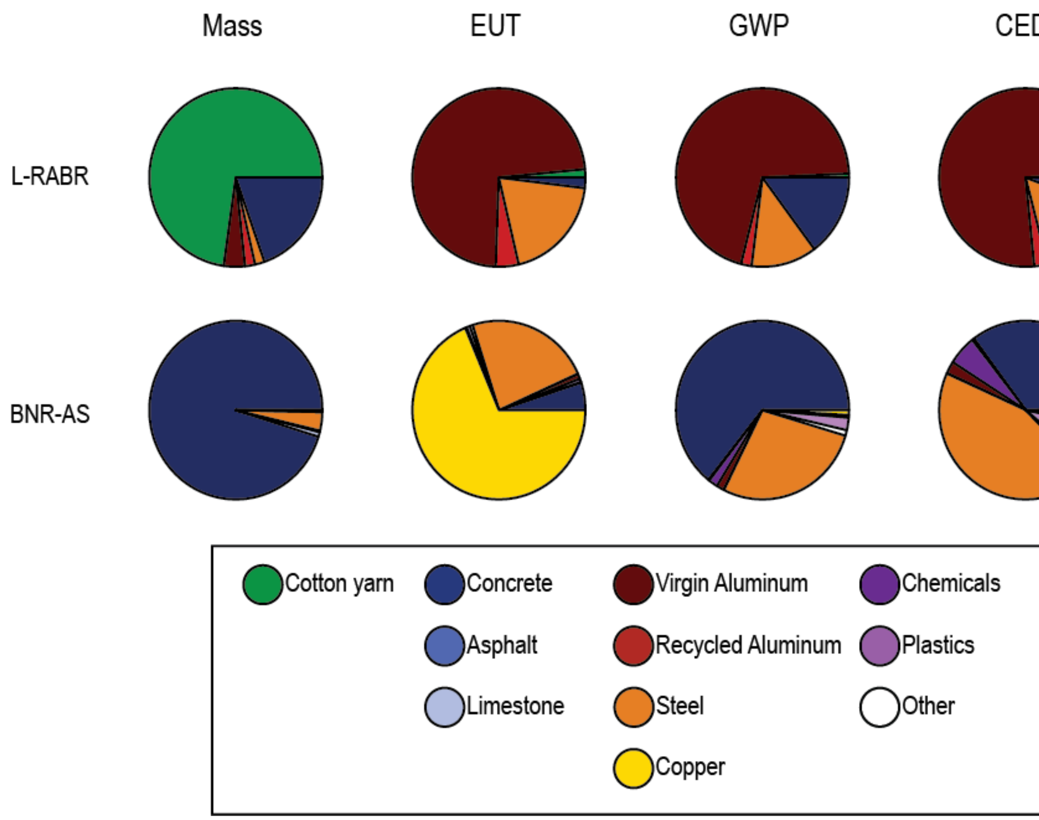


Figure 4.15: Mass of materials and contributions of each to construction stage impacts for both upgrade scenarios and all impact categories.

The construction stage impact assessment of the BNR-AS scenario shows that among the materials used in construction of more conventional plants concrete, steel, and copper contribute most significantly to the life cycle impacts. Concrete and steel are the major contributors to GWP and CED, whereas copper and steel have a large eutrophication impact (Figure 4.15). The contribution of concrete would increase if other excluded gate-to-gate processes were included, as it comprises the majority of the mass used in construction of the activated sludge system. This information is useful in trying to minimize the environmental burden (i.e. maximize the net environmental benefit) brought on by the wastewater treatment industry.

Algal Biomass Production

In addition to the impacts and benefits considered in this assessment, the L-RABR systems would produce a potentially valuable resource: algal biomass. Assuming 20 mg/L-m^2 (Christenson and Sims, 2012), the L-RABR could produce around 17,000 kg of dry biomass per year. The algal biomass could then be used in a number of applications such as energy or bioplastics production, among other uses. It is predicted that the gross energy content, based on an average of 18 MJ/kg biomass (Ferrell and Sarisky-Reed, 2010), would be around 85,000 kWh/year. Alternatively, if used to replace fossil-carbon based industrial chemicals, assuming 0.25 g solvent/g biomass (Ellis et al., 2012), 4,250 kg solvent/year could be produced including acetone, butanol, and ethanol; residuals from solvent production could still be used to produce energy. It is expected that if the end-use of the algal biomass was included in this analysis, the benefits of the L-RABR system would be even greater than those reported here. Even assuming the worst case scenario for the algal biomass where it is landfilled, the L-RABR scenario only produces 62% by mass of the biosolids sludge landfilled in the BNR-AS scenario (calculated for the design year) and thus would emit less GHGs after disposal.

Conclusions

RABRs present an interesting opportunity for existing lagoon wastewater treatment systems that necessitate nutrient removal. Modeling based on lab- and pilot-scale data show that combined lagoon-RABR systems are able to achieve reliable nutrient removal with only marginal increases in electricity use beyond the lagoons alone. These systems impart less environmental impacts over all life cycle stages than the more conventional upgrade approach to under-performing wastewater treatment lagoons: BNR activated sludge systems. The results of this study can be used by engineers, policy makers, and operators to identify trade-offs for upgrading outdated lagoon systems across the United State and abroad. It is recommended that site-specific conditions be considered and pilot-scale RABR tests be performed when necessary to confirm the applicability of the L-RABR model to the local water and weather conditions; however, much of the data on construction and operation of these systems is not location dependent and thus can be used to guide decisions about the future direction of lagoon wastewater treatment systems. Additionally, the uncertainty observed in the BNR-AS models point to the importance of considering variability in influent quality when design assessment is performed. The method for performing Monte Carlo analysis with BioWin models used in this study is an important contribution to a wide variety of wastewater-focused systems analyses and should be employed in future studies.

CHAPTER FIVE

CONCLUDING REMARKS

This work presented three studies to improve the understanding of the impacts of integrating algaculture in a wastewater treatment context. The first study used theoretical models of algal and wastewater processes to compare the performance of various hybrid algaculture/activated sludge treatment systems in a life cycle assessment framework. The second study addressed algal biomass harvesting which was outside of the system boundaries of the first study but is likely to have an influence on the success of algaculture systems and which have not yet been studied sufficiently in literature. This study used laboratory investigations to understand how nutrient limitation that results from algal nutrient removal might hinder the implementation of membrane separation processes for algaculture in wastewater treatment. The third study improved on methods from the first study using lab- and pilot-scale data in conjunction with life cycle modeling techniques to compare the performance of hybrid algal biofilm/lagoon and conventional activated sludge treatment systems. The key findings are summarized below.

Integrating algaculture into small wastewater treatment plants: Process options and life cycle impacts

- Incorporating algaculture processes at small wastewater treatment plants with available land can improve the sustainability of treatment processes.
- Primary algal nutrient removal is a promising integration approach due to reductions in operational energy and biosolids production.

- Additional processes that would be required (such as primary sedimentation or algal harvesting) should be considered.
- Improvements in effluent quality and efficiency over conventional treatment strategies can provide an innovative way to satisfy the growing interest in energy and resource recovery in the wastewater industry.

Effects of nitrogen limitation and culture density in algae systems using microfiltration

- The hypothesis that nitrogen limitation promotes fouling in algae culture/microfiltration systems was supported by this work.
- Nitrogen stress and subsequent accumulation of carbon-rich intracellular metabolites that are then excreted from the cell due to concentration-driven diffusion causes fouling.
- Dilution of culture density can also promote this concentration-driven diffusion and result in fouling.
- When cells are transitioned from high to low nitrogen environments, this phenomenon can be exacerbated because of the higher cell count per unit biomass achieved in the nutrient-rich setting.
- The impact of the relationship of culture conditions and fouling propensity should be considered when implementation of membrane filtration is used for algal WWT or biofuels applications.

Comparative life cycle assessment of nutrient removal options for existing lagoon systems: Attached growth

- Results show that combined lagoon/RABR systems improve eutrophication impacts relative to the existing lagoons, more so than the BNR system.
- The increase in global warming potential and cumulative energy demand which resulted from increased energy and chemical use as well as construction of the upgraded systems was less for the lagoon/RABR system than for the BNR treatment plant.
- Algaculture systems may be applied when land intensive wastewater treatment strategies are acceptable, as is the case with existing lagoons.
- The majority of the data and background modeling for this study is applicable to a wide variety of locations, though some local conditions can be considered and pilot-scale RABR tests be used to confirm the applicability of the L-RABR model to a more specific system of interest.
- The methods used for performing Monte Carlo analysis with BioWin models show the importance of considering variability in influent quality in wastewater modeling and represent an important contribution to the field; these methods should be employed in future studies.

APPENDICES

Appendix A

Supplementary Information for Chapter Two: Integrating algaculture into small wastewater treatment plants: Process flow options and life cycle impacts

Nutrient removal values were generated using the gamma distribution, where alpha and beta (shape and rate parameters, respectively) were set to best fit the data reported in literature. First, histograms of data obtained from the literature were plotted for each wastewater type (primary treated, secondary treated, and sidestream wastewaters), and the percent of instances when removal was >95% and 75-95% were determined. The function 1-GAMMA.INV(RAND(),α,β) in Excel was used to generate 1000 values of removal, and alpha and beta parameters were varied until the percent of instances when removal was >95% and 75-95% matched that of the literature data. Histograms for data and model are shown in Figures A1-A6. Alpha and beta parameters and resulting nutrient removal values for TANR, PANR, and SANR models are shown in Tables A1-A3.

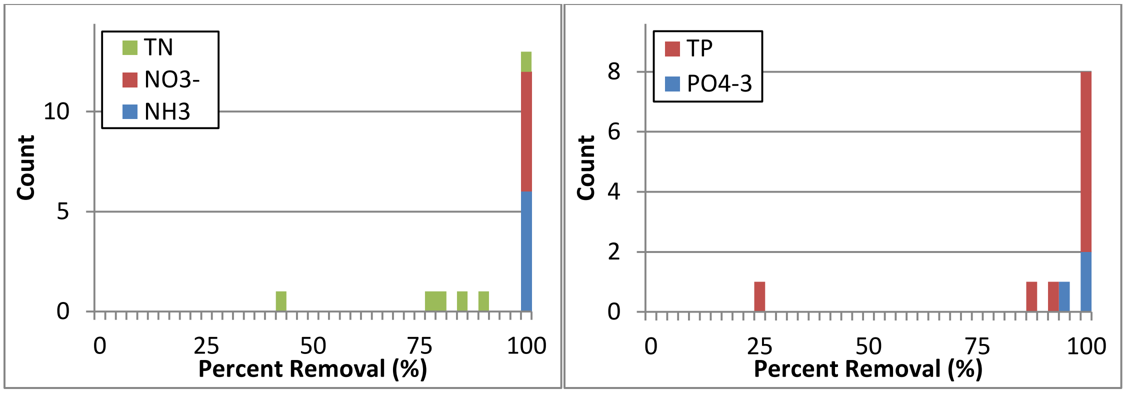


Figure A1: Histograms of nitrogen (left) and phosphorus (right) removal reported in literature for secondary treated wastewater.

Table A1: Final gamma distribution parameters and resulting removal values for TANR model.

	TN	TP
alpha	0.75	0.75
beta	5	5
Mean	96.3	96.4
Max	100.0	100.0
Min	69.6	73.3
St Dev	4.2	4.1

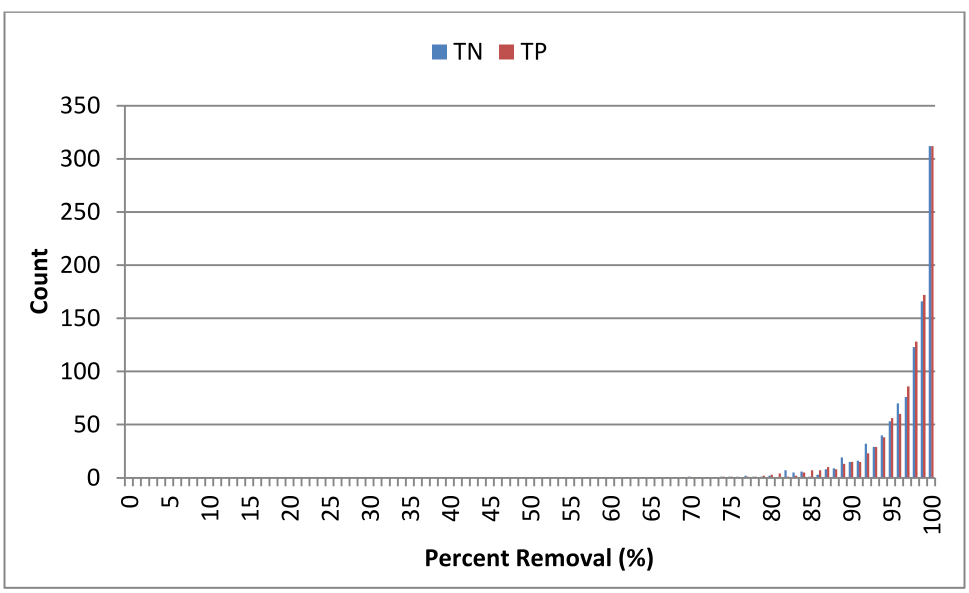


Figure A2: Histogram for TN and TP removal values used in TANR model.

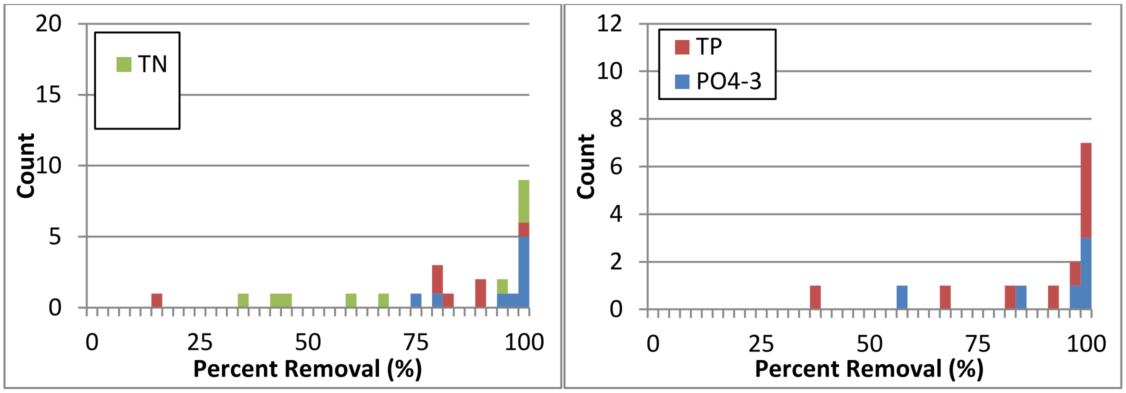


Figure A3: Histograms of nitrogen (left) and phosphorus (right) removal reported in literature for primary treated wastewater.

Table A2: Final gamma distribution parameters and resulting removal values for PANR model.

	TN	TP
alpha	1	0.75
beta	10	6
Mean	89.9	95.5
Max	100.0	100.0
Min	40.1	47.8
St Dev	9.7	5.4

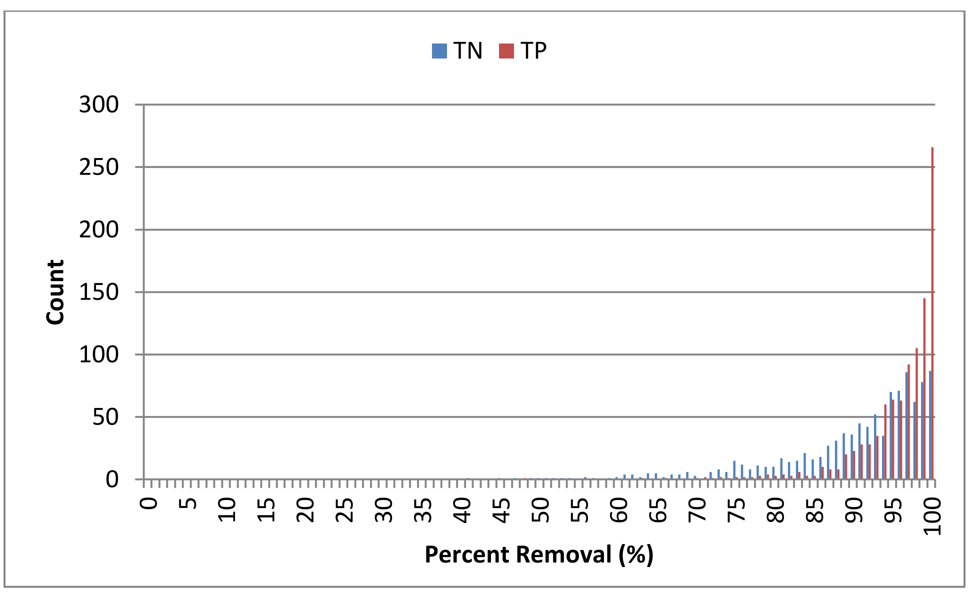


Figure A4: Histogram for TN and TP removal values used in PANR model.

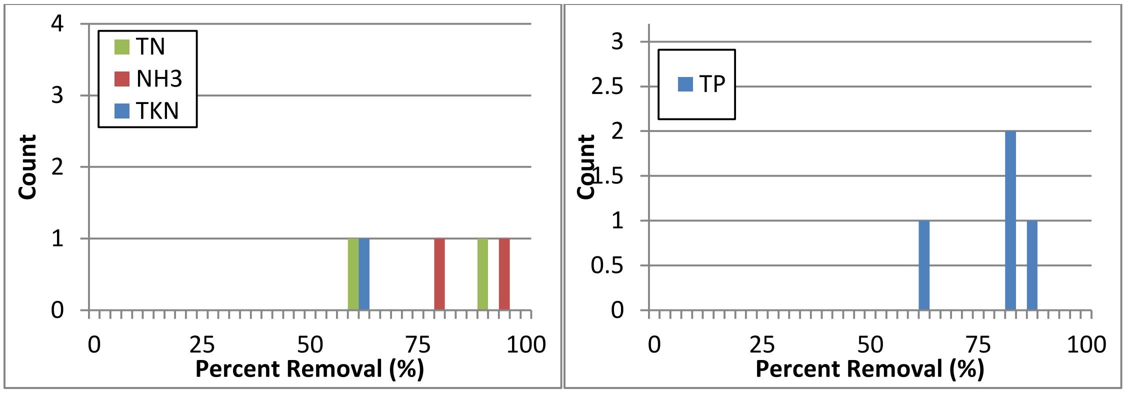


Figure A5: Histograms of nitrogen (left) and phosphorus (right) removal reported in literature for sidestream wastewater.

Table A3: Final gamma distribution parameters and resulting removal values for SANR model.

	TN	TP
alpha	3	3
beta	8	6
Mean	76.0	82.3
Max	99.3	98.6
Min	10.5	39.8
St Dev	14.0	9.9

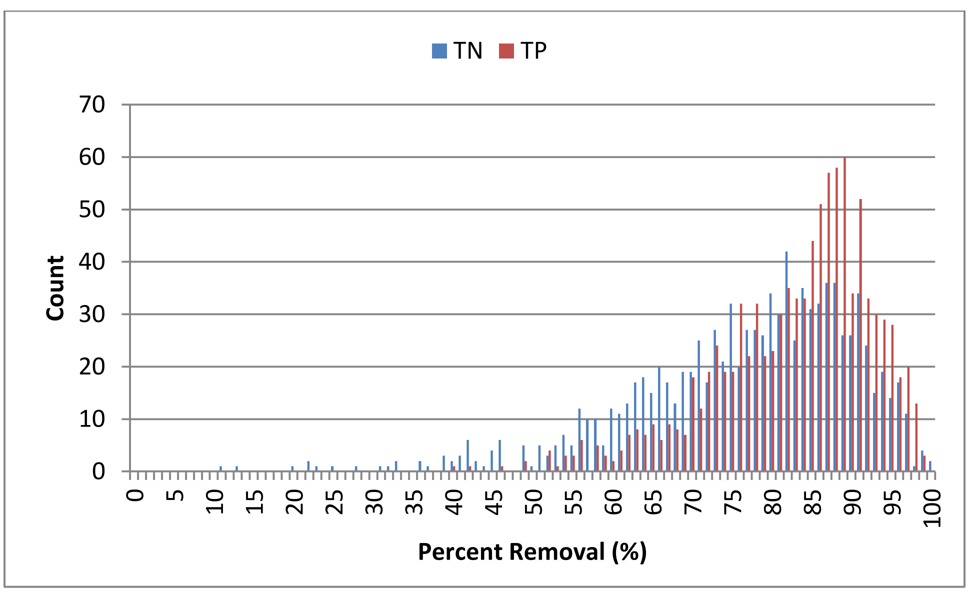


Figure A6: Histogram for TN and TP removal values used in SANR model.

Sensitivity of the algaculture models to each parameter in the model (TN removal, TP removal, and stoichiometric coefficients for H, P, C, O, and N of algal biomass) was determined using Monte Carlo analysis, where parameters were varied one at a time and impacts to algal biomass production and N and P uptakes were determined. Tornado plots for this analysis are shown in Figures A7-A9. Each bar is centered on the mean of the distribution and extends two standard deviations from top to bottom.

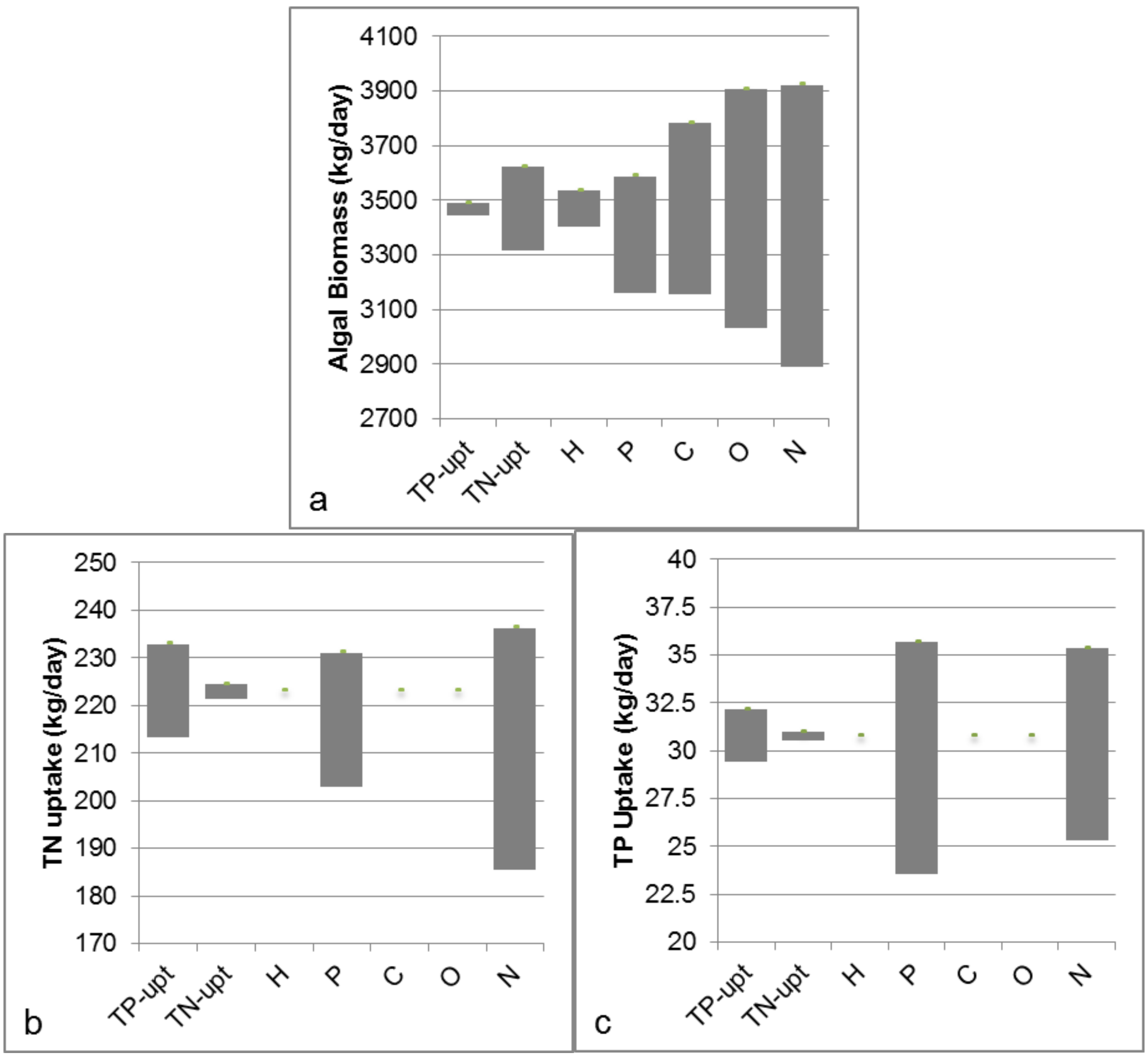


Figure A7: Tornado plots of the sensitivity of (a) algal biomass production, (b) nitrogen uptake, and (c) phosphorus uptake to seven input parameters in the TANR model.

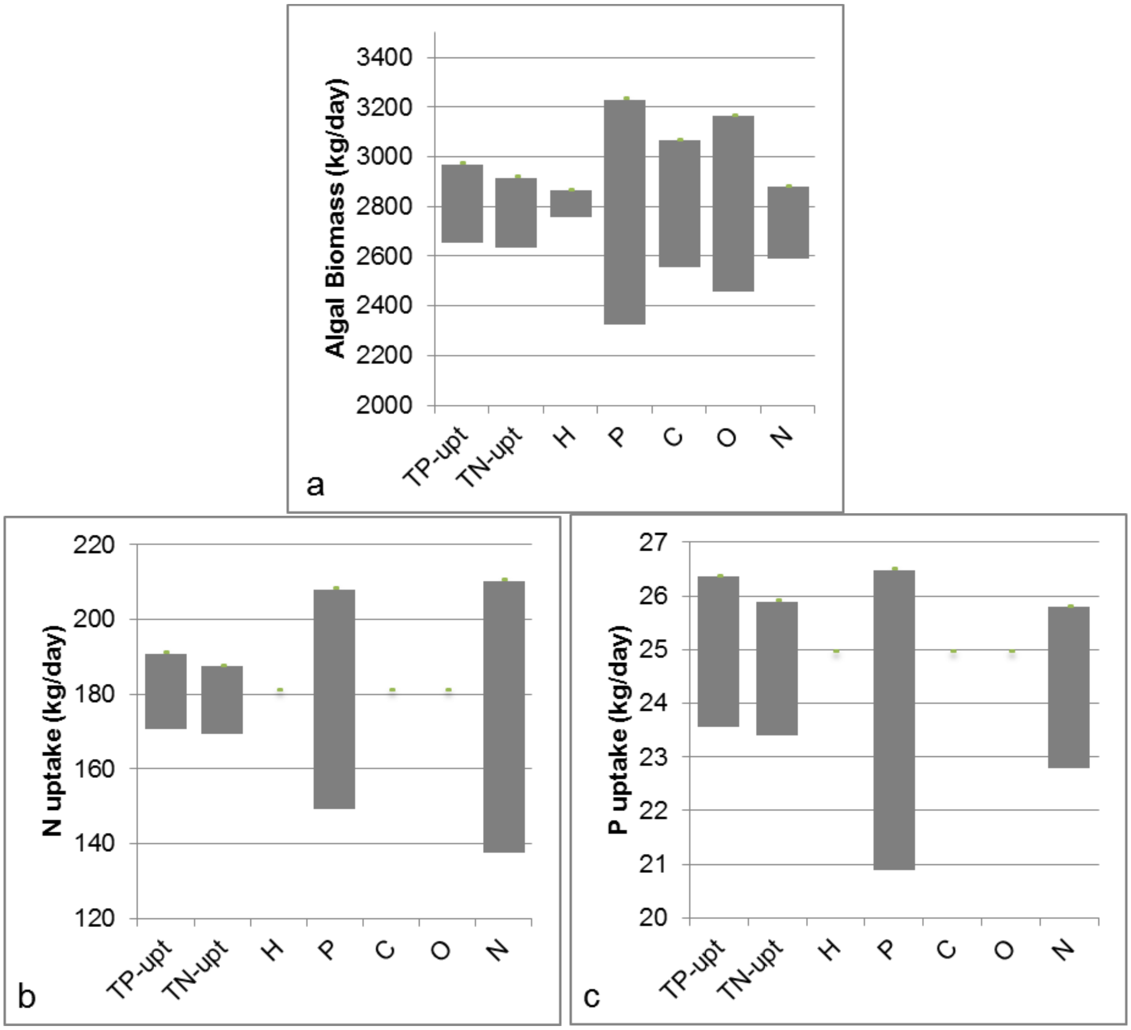


Figure A8: Tornado plots of the sensitivity of (a) algal biomass production, (b) nitrogen uptake, and (c) phosphorus uptake to seven input parameters in the PANR model.

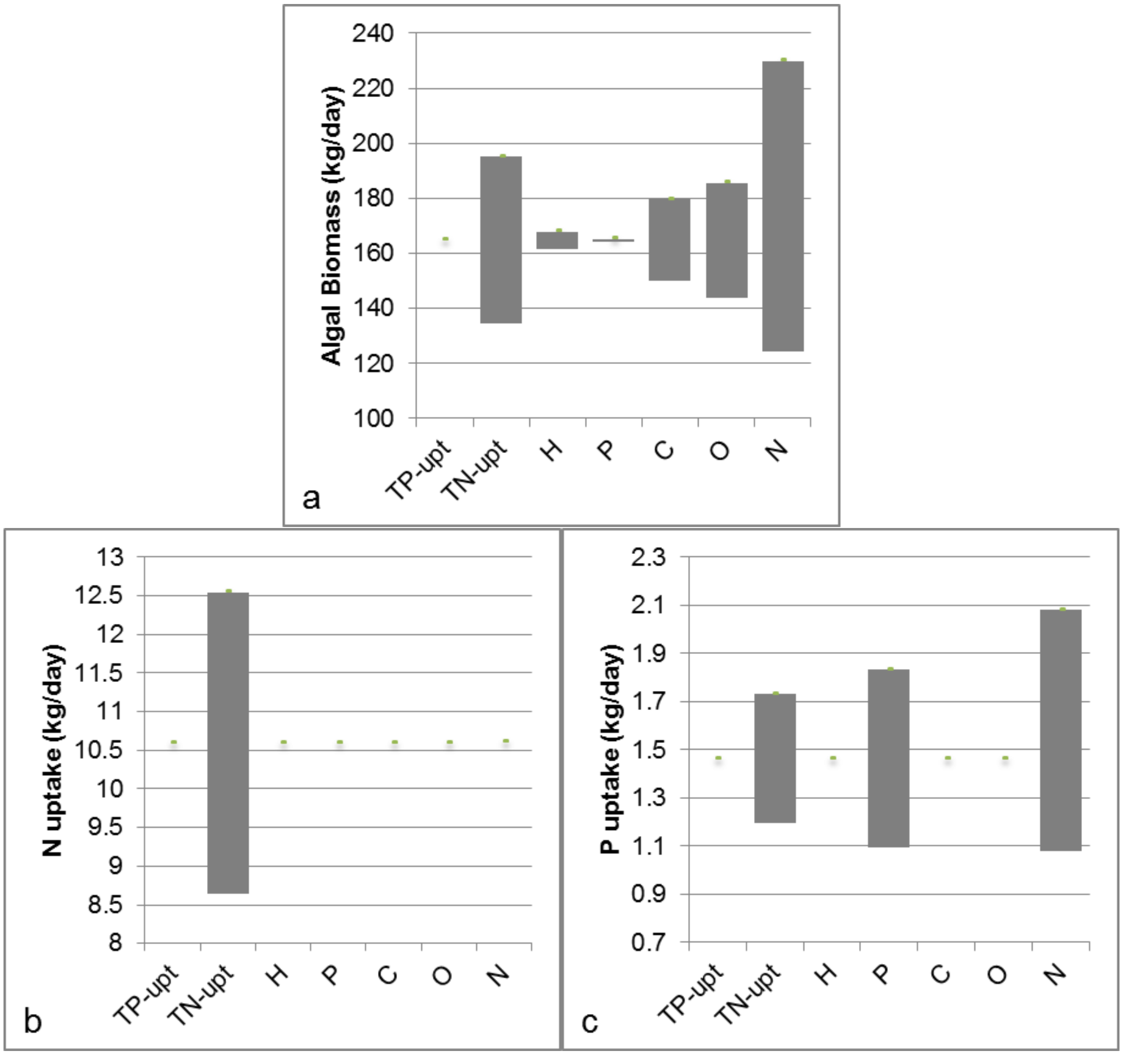


Figure A9: Tornado plots of the sensitivity of (a) algal biomass production, (b) nitrogen uptake, and (c) phosphorus uptake to seven input parameters in the SANR model.

Appendix B

Supplementary Information for Chapter Three: Effects of nitrogen limitation and cultures density in algae systems using microfiltration

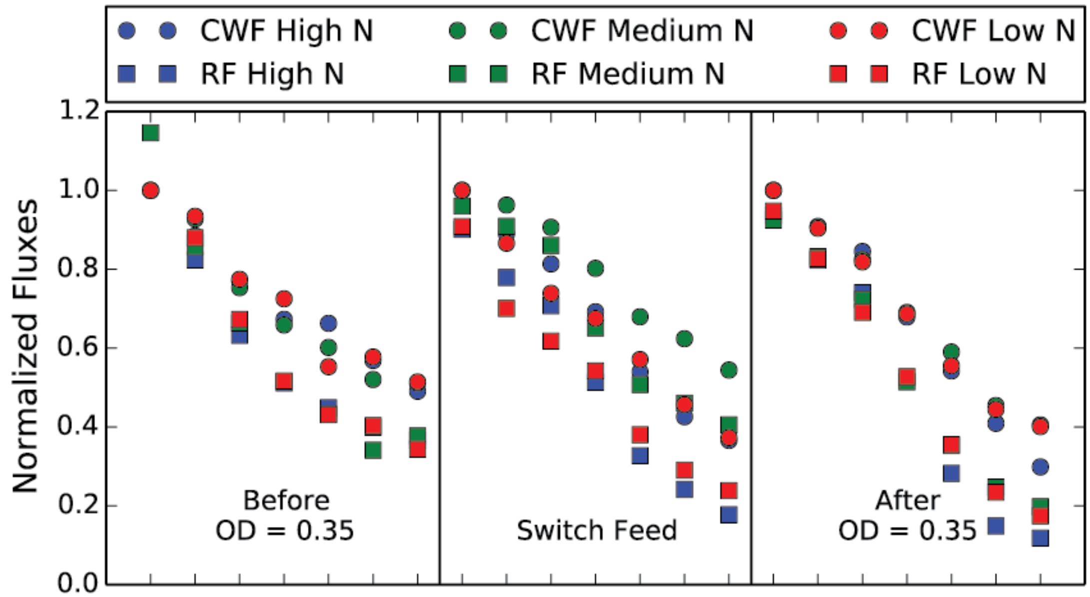


Figure B1: Normalized flux decline curves for three membrane coupons before, during, and after feed nitrogen concentrations were switched, which occurred after the first data point in the panel labeled "Switch Feed".

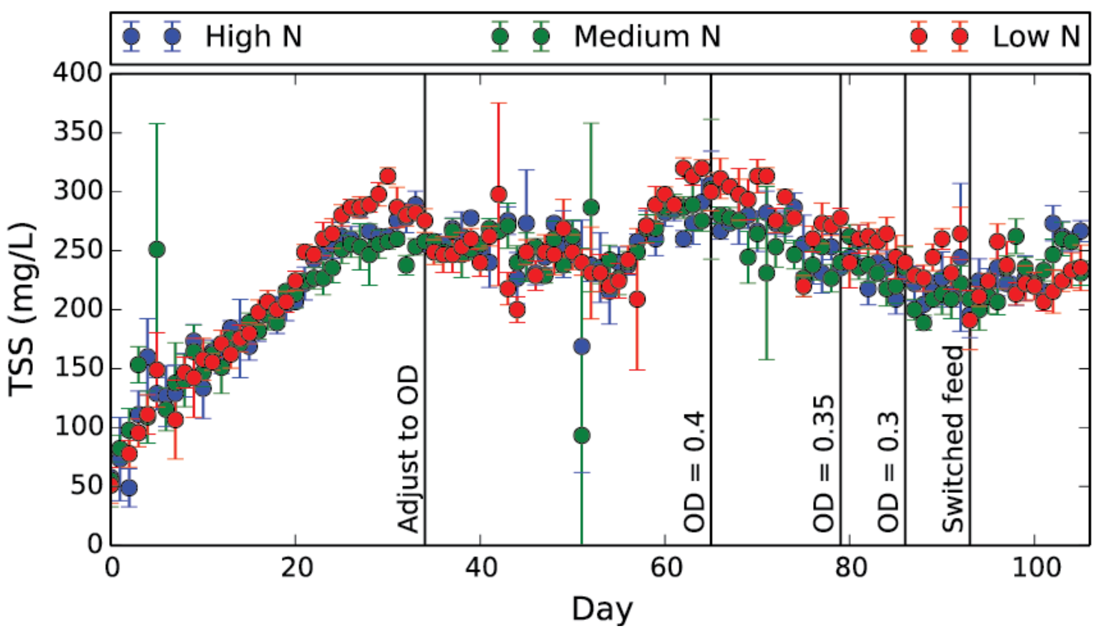


Figure B2: TSS measurements over the entirety of the 107 day experiment.

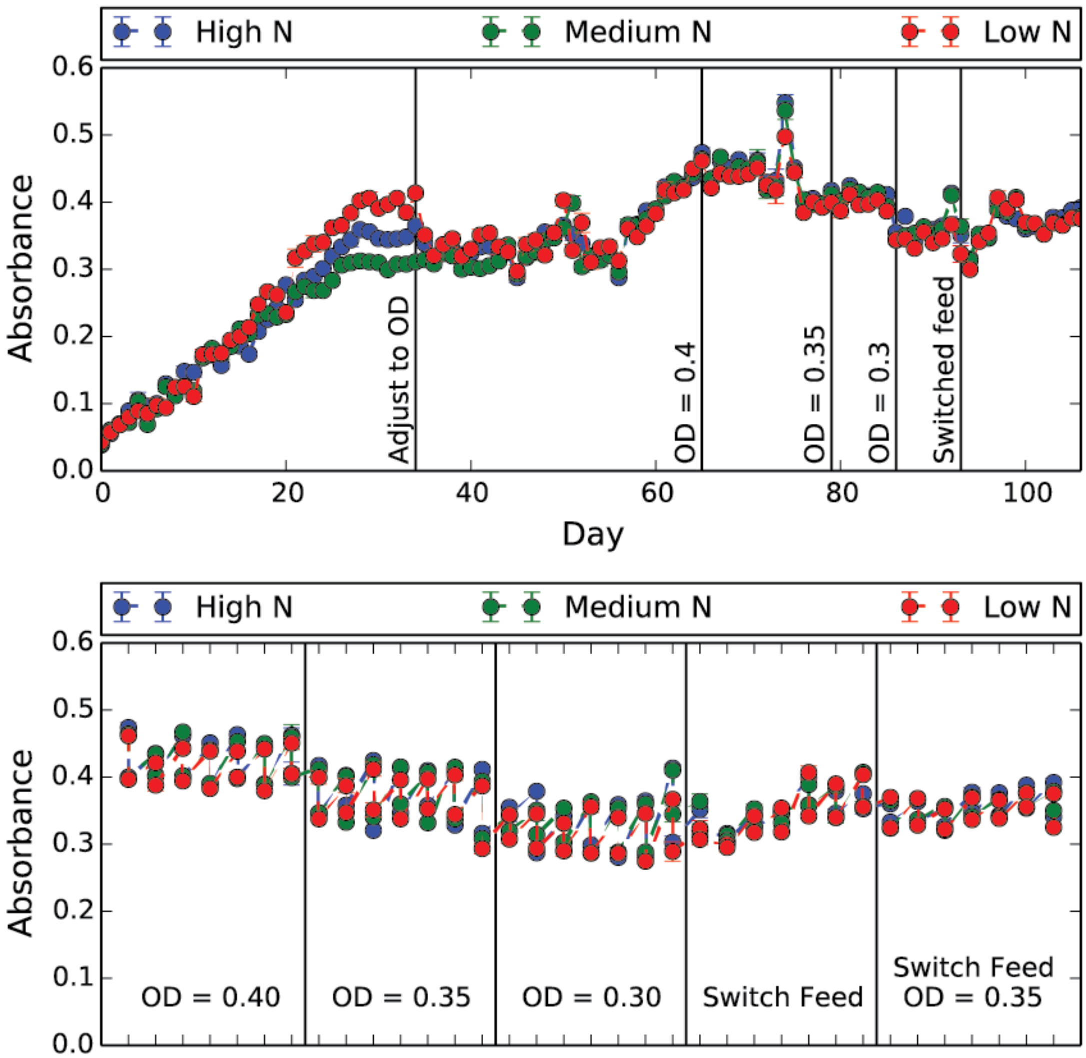


Figure B3: Optical density measurements, used to determine culture density in real time, for the entire 107 day experiment (top) and for the membrane coupons for which filtration data is shown (bottom).

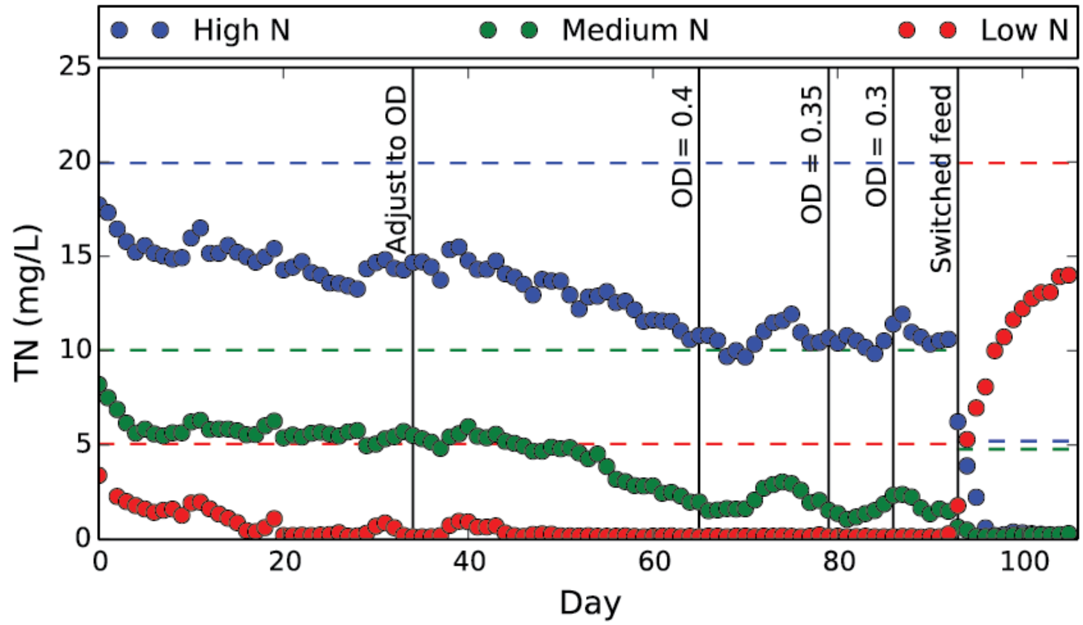


Figure B4: Total nitrogen concentrations over the entire 107 day experiment. Dotted line represent feed concentations.

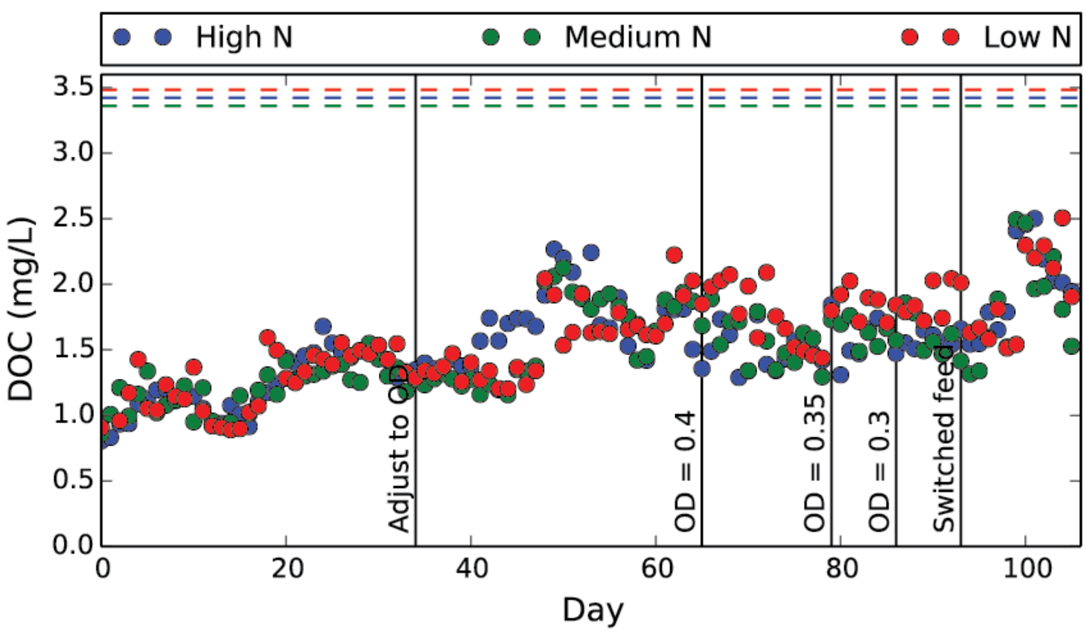


Figure B5: Dissolved organic carbon concentrations over the entire 107 day experiment.

Appendix C

Supplementary Information for Chapter Four: Comparative life cycle assessment of nutrient removal options for existing lagoon systems: Attached growth algae retrofit versus greenfield activated sludge construction

1 – Historical Lagoon Data

Data collected at the existing lagoon treatment plant in Logan, Utah between 2010 and 2013 were used in this study. Daily volumetric flows in million gallons per day (MGD) for influent and effluent and weekly measurements of biological oxygen demand (BOD), total suspended solids (TSS), total phosphorus (TP), and ammonia concentrations in milligrams per liter were provided. In addition, monthly energy use data were provided for 2010 through 2013. Descriptive statistics for these data are provided in Tables C1-C6a. The energy use in 2013 did not follow the trends seen in years 2010-2012 (Figure C1). This difference was attributed to a number of factors including colder winter temperatures, resulting in more frequent use of surface aerators to break up ice on the lagoons, as well as other activities at the treatment plant, including various pilot-scale test units for effluent polishing. Therefore, 2013 energy data was excluded (Table C6b) from extant energy analysis, as described later.

Table C1: Volumetric flow data (2010-2013), reported in million gallons per day (MGD)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	10.3	10.1	12.1	12.8	13.8	16.3	16.7	15.4	14.1	11.9	10.1	10.4
St. Dev.	1.7	0.8	2.5	4.6	2.5	1.3	1.2	1.3	1.1	1.5	1.0	2.9
Median	9.4	9.9	11.0	10.9	13.9	16.0	16.7	15.4	14.1	11.6	10.1	9.7
Min	8.1	8.5	9.2	9.2	9.2	14.0	14.3	13.0	11.4	9.6	7.5	6.8
Max	15.5	11.9	19.3	36.4	26.1	20.1	19.8	22.2	16.7	18.0	12.9	24.8
Count	124	113	124	120	124	120	124	124	120	124	120	124

Table C2: Monthly BOD data (2010-2013), reported in mg/L

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	14.2	22.1	25.1	15.1	12.8	21.2	13.6	7.5	8.3	5.0	13.7	16.4
St. Dev.	7.4	6.2	6.8	5.3	3.2	16.1	9.2	5.1	8.4	7.9	8.7	8.1
Median	16.0	21.0	24.0	14.0	13.0	18.0	18.0	6.8	2.5	2.5	15.0	17.0
Min	2.5	12.0	16.0	7.0	8.0	2.5	2.5	2.5	2.5	2.5	2.5	2.5
Max	28.0	36.0	44.0	27.0	17.0	45.0	23.0	14.0	24.0	37.0	29.0	33.0
Count	17	17	18	9	4	14	5	4	5	18	17	18

Table C3: Monthly TSS data (2010-2013), reported in mg/L

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	35.2	40.7	43.8	25.4	21.5	17.6	18.7	15.3	11.3	11.7	30.5	35.9
St. Dev.	7.2	15.4	15.3	15.5	11.7	13.9	6.6	7.1	3.6	6.2	6.6	10.2
Median	36.0	41.0	50.0	21.0	17.5	14.0	20.0	15.0	12.0	10.5	30.0	36.5
Min	20.0	9.0	23.0	12.0	8.0	2.0	7.0	2.0	5.0	2.0	20.0	10.0
Max	47.0	77.0	76.0	80.0	42.0	51.0	32.0	28.0	17.0	24.0	45.0	59.0
Count	17	17	18	16	18	18	17	17	18	18	17	18

Table C4: Monthly total phosphorus data (2010-2013), reported in mg/L

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	3.1	3.2	3.0	3.0	3.1	3.8	3.2	2.6	2.6	2.7	2.5	2.7
St. Dev.	0.4	0.6	0.8	0.6	0.5	0.8	0.6	0.3	0.4	0.3	0.3	0.2
Median	3.2	2.9	2.9	3.1	3.0	3.8	3.2	2.5	2.7	2.8	2.4	2.7
Min	2.5	2.4	2.0	2.1	2.3	2.8	2.4	2.1	1.9	2.2	2.1	2.2
Max	3.7	4.4	4.3	3.9	4.0	6.2	4.8	3.1	3.2	3.2	3.0	3.1
Count	17	17	18	16	18	18	17	17	18	18	17	18

Table C5: Monthly ammonia data (2010-2013), reported in mg/L

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	13.0	15.4	14.3	12.2	7.4	6.3	3.7	2.7	4.8	6.9	6.1	9.1
St. Dev.	2.8	3.8	4.1	4.4	4.1	4.8	3.7	1.8	1.6	2.1	2.7	3.7
Median	11.9	15.4	14.6	12.1	8.4	6.8	1.0	2.8	4.6	7.9	5.4	9.9
Min	8.7	9.7	7.3	6.0	0.1	0.2	0.2	0.4	0.9	3.9	1.7	2.0
Max	17.6	22.1	20.3	19.3	12.7	14.4	10.3	5.9	7.7	10.0	10.5	14.0
Count	17	17	18	16	18	18	17	17	18	18	17	18

Table C6a: Monthly energy consumption (2010-2013), reported in MWh

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	346.2	285.3	287.5	360.0	392.6	266.6	270.6	267.7	258.9	337.5	296.0	250.2
St. Dev.	309.0	232.7	209.7	197.5	153.7	49.6	75.4	98.7	87.5	107.9	91.5	129.2
Median	184.3	165.5	176.3	316.7	350.5	255.1	258.5	256.6	251.4	326.2	319.8	192.6
Min	136.6	124.0	147.6	137.3	237.0	219.6	183.3	146.7	144.8	207.9	154.7	148.2
Max	879.6	686.4	649.8	669.4	632.4	336.7	382.1	410.8	388.0	489.7	389.7	467.4
Count	4	4	4	4	4	4	4	4	4	4	4	4

Table C6b: Monthly energy consumption (2010-2012), reported in MWh

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	168.4	151.7	166.7	256.9	312.7	282.3	299.7	308	296.9	360.1	264.8	177.8
St. Dev.	29.2	26.7	17.2	97.2	77.1	48	64.7	80.5	66.6	116.1	85.1	35.9
Median	161.4	143.1	163.2	257.9	282.5	290.2	292.9	298.9	272.1	382.7	277.7	157
Min	136.6	124	147.6	137.3	237	220	224.1	214.2	230.6	207.9	154.7	148.2
Max	207.2	187.8	189.4	375.4	418.5	336.7	382.1	410.8	388	489.7	362	228.3
Count	3	3	3	3	3	3	3	3	3	3	3	3

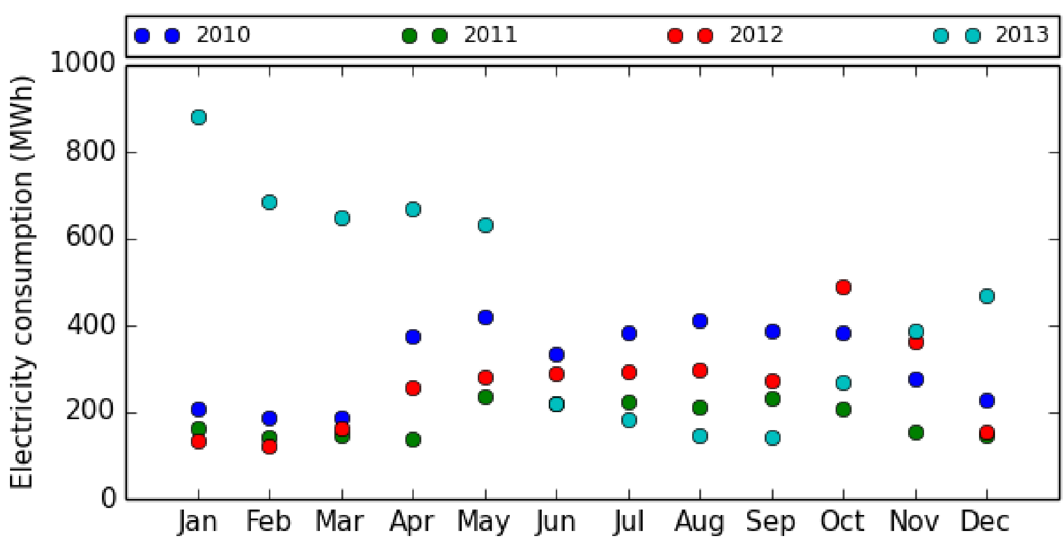


Figure C1: Monthly energy use (2010-2013).

2 – Design Limit Calculation

Because phosphorus is not regulated on a concentration basis, the influent flow rate was used to determine the approximate monthly concentration necessary to meet the limits, as follows:

$$Limit, Concentration \left[\frac{mg}{L}\right] = \frac{Limit,\ Cumulative\ [kg] \cdot 1000 \left[\frac{g}{kg}\right] \cdot 1000 \left[\frac{mg}{g}\right]}{Season\ Length\ [days] \cdot Flow \left[\frac{gallons\ \times 10^6}{day}\right] \cdot 3.73 \left[\frac{liters}{gallon}\right]}$$

3 – L-RABR Design and Modeling

Design of the RABR installation into lagoon pond D were based on staged design of rotating biological contactor (RBC) systems, as described in Grady et al. (2011). Dimensions for the RABR unit footprint (1.8 m wide by 2.4 m long) were obtained from Christenson and Sims (2012).

A combination of lab- and pilot- scale data regarding nutrient removal by RABR systems were obtained from Utah State University. Removal rate (in milligrams per liter per day) was plotted against nitrogen concentration. A linear trend was observed (R2 = 0.943) within the range of nitrogen concentrations relevant in a wastewater context (<50 mg N/L). The slope was used as the first-order reaction rate (0.461 d-1) in the RABR treatment model (Figure C3). For uncertainty analysis, values for the first-order nitrogen removal rate were randomly distributed within ± 25% of 0.461 d-1. The removal rate of phosphorus was not observed to be concentration dependent. Therefore average removal rate (0.379 mg L-1d-1) was used. For uncertainty analysis, a triangular distribution was used to generate 1000 values of the zero-order removal rate (Figure C4). Low removal rates were correlated to higher algal biofilm ages (>12 days), therefore, the harvesting of biomass is suggested at or before 12 days of growth.

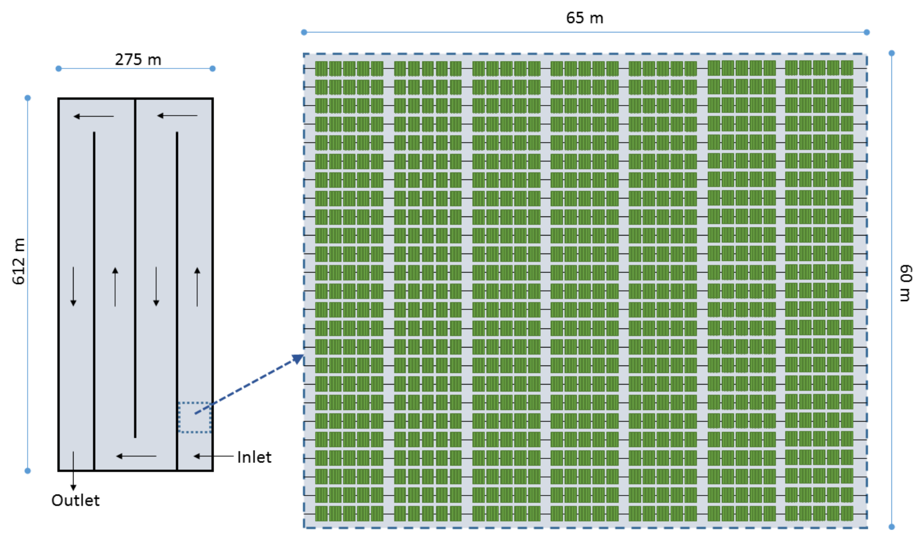


Figure C2: Plan view of pond D with channels (left) and stage configuration (right) for RABR installation.

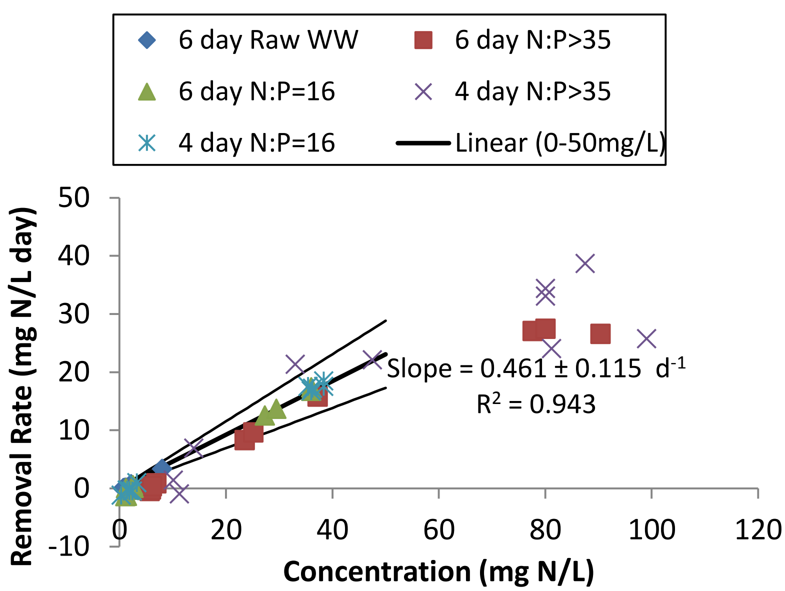


Figure C3: Nitrogen removal data used to determine first order removal rate for RABR model. Different marker types represent different experimental tests performed at Utah State University.

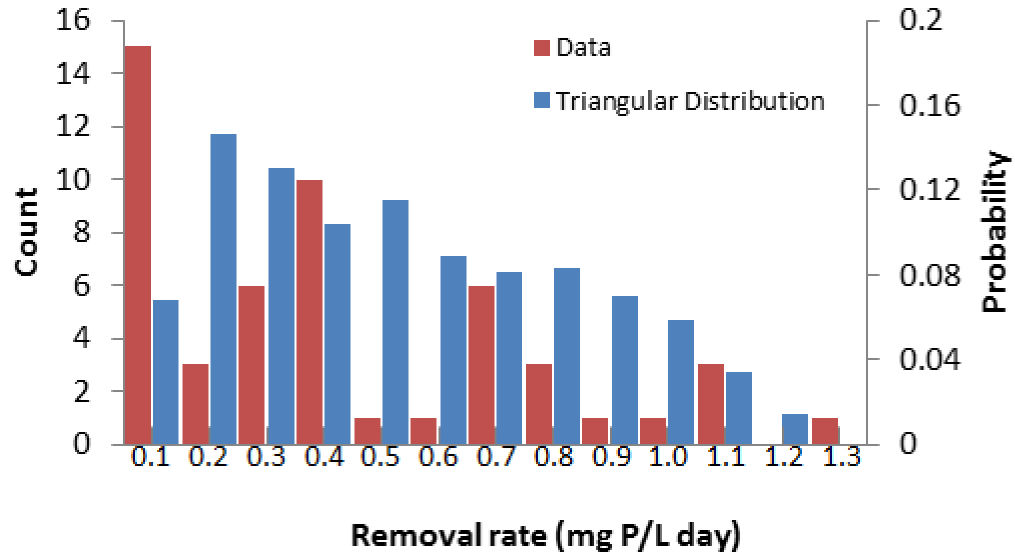


Figure C4: Phosphorus removal data used to determine zero order removal rate for RABR model.

The number of stages required per month was determined using design influent conditions (Table C7) based on achieving ammonia and phosphorus limits in the effluent; design values for number of stages was set to meet both limits when possible (Table C8). When ammonia removal was the determining factor, a buffer stage was added to ensure compliance. No buffer stage was added for months when phosphorus was the determining factor because it was assumed alum could be used, if necessary, to maintain compliance. Reaching effluent phosphorus limit is not achievable in June with the L-RABR system as modeled and requires significantly more stages than other months in July. This is due to high volumetric flow rates and high TP concentrations in these months. Therefore, the design number of stages was selected based on the number of stages used May and August so large shifts in the number of stages would not necessary month to month. Alum use was calculated when phosphorus limits were not achieved using the parameters in Table C9.

Sludge produced as a result of alum use estimated stoichiometricly. Ninety percent phosphorus removal was assumed for the alum dose modeled (G; Tchobanoglous et al., 2003).

Table C7: RABR design influent conditions. Flow values are in MGD; all other values are in mg/L.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Flow	10.3	10.1	12.1	12.8	13.8	16.3	16.7	15.4	14.1	11.9	10.1	10.3
BOD	14.2	22.1	25.1	15.1	12.8	21.2	13.6	7.5	8.3	5	13.7	14.2
TP	3.1	3.2	3	3	3.1	3.8	3.2	2.6	2.6	2.7	2.5	3.1
NH_3	13	15.4	14.3	12.2	7.4	6.3	3.7	2.7	4.8	6.9	6.1	13
TSS	35	41	44	25	22	18	19	15	11	12	31	35

Table C8: RABR Stages used during each month based on ammonia-N limit, P limit, and used in the final design. NA= not achievable.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
NH_3	14	15	17	16	11	22	15	10	8	11	9	11
TP	13	13	17	19	25	NA	35	22	19	15	6	9
Design	**15**	**16**	**18**	**19**	**25**	**24**	**23**	**22**	**19**	**15**	**10**	**12**

Table C9: Parameters for in alum use calculations.

Parameter	Value and Units
Al:P ratio	2 mol Al/mol P
Alum formula	$Al_2(SO_4)_3 \cdot 18\ H_2O$
Percent Solution	48 %
Density	1.2 kg/L

$$Al^{3+} + PO_4^{3-} \leftrightarrow AlPO_4$$

$$2Al^{3+} + 3OH^{-} \leftrightarrow Al(OH)_3$$

4 – BNR-AS Design and Modeling

Information regarding the design of the BNR-AS scenario was obtained from the City of Logan's Wastewater Treatment Master Plan (Carollo Engineers, 2013) and from direct correspondence with Carollo (Tables C10-C11).

Table C10: Design parameters for BNR-AS system

Parameter	Value	Unit
# of replicate reactors	6	-
# of replicate clarifiers	6	-
Basin depth	15	ft
SRT	20	days
MLR	4	Q
O_2 transfer (surface aer.)	3.5	lbs O_2/hp-hr

Table C11: Volume of each zone, totals and per reactor, in million gallons

	Total volume	Volume/reactor
Anaerobic	1.0	0.167
Anoxic	3.0	0.5
Aerobic	8.9	1.5

The BNR-AS scenario was modeled in BioWin (EnviroSim, v 4.0) using one treatment train and because not all six reactors would be necessary at all flows, the pertinent data was then multiplied by the number of reactors required to handle the influent volumetric flow. Model parameters (Tables C12-C13) were adjusted to correspond with typical municipal wastewater treatment activated sludge plants (C. P. Leslie Grady et al., 2011).

Influent concentrations values were then converted into units consistent with the BioWin model parameters (BOD → COD, TSS → ISS, NH_3 → TKN; Table S14). Total COD was assumed to be 2.1 times BOD (C. P. Leslie Grady et al., 2011). Ammonia values were converted to total Kjeldahl nitrogen (TKN) using molecular weights of ammonia and nitrogen, and assuming 0.75 mg NH_3-N per mg TKN (a BioWin default). Inert suspended solids (ISS) were assumed to be 15% of TSS. Other influent parameters were left as BioWin defaults (0 mg NO_3/L, pH 7.3, 6 mmol alkalinity/L, 80 mg Ca/L, 15 mg Mg/L, 0 mg DO/L).

Table C12: Non-default stoichiometric parameters used in BioWin (all other parameters left as default)

Common	Default	Value
Biomass volatile fraction (VSS/TSS)	0.92	0.85
Endogenous residue volatile fraction (VSS/TSS)	0.92	0.85
N in endogenous residue [mgN/mgCOD]	0.07	0.06
Ammonia Oxidizing Bacteria	**Default**	**Value**
N in biomass [mgN/mgCOD]	0.07	0.086
Nitrite Oxidizing Bacteria	**Default**	**Value**
N in biomass [mgN/mgCOD]	0.07	0.086
Ordinary Heterotrophic Organisms	**Default**	**Value**
Yield (aerobic) [-]	0.666	0.6
N in biomass [mgN/mgCOD]	0.07	0.086

Table C13: Non-default kinetic parameters used in BioWin (all other parameters left as default)

Ammonia Oxidizing Bacteria	Default	Value
Max. spec. growth rate [1/d]	0.9	0.768
Substrate (NH4) half sat. [mgN/L]	0.7	1
Aerobic decay rate [1/d]	0.17	0.096
Anoxic/anaerobic decay rate [1/d]	0.08	0.096
Nitrite Oxidizing Bacteria	**Default**	**Value**
Max. spec. growth rate [1/d]	0.7	0.768
Aerobic decay rate [1/d]	0.17	0.096
Anoxic/anaerobic decay rate [1/d]	0.08	0.096
Ordinary Heterotrophic Organisms	**Default**	**Value**
Max. spec. growth rate [1/d]	3.2	6
Substrate half sat. [mgCOD/L]	5	20
Anoxic growth factor [-]	0.5	0.8
Aerobic decay rate [1/d]	0.62	0.408
Anoxic decay rate [1/d]	0.233	0.408
Hydrolysis rate [1/d]	2.1	2.208
Hydrolysis half sat. [-]	0.06	0.15
Anoxic hydrolysis factor [-]	0.28	0.4
Adsorption rate of colloids [L/(mgCOD d)]	0.15	0.1608
Ammonification rate [L/(mgN d)]	0.04	0.04
Switches	**Default**	**Value**
Aerobic/anoxic DO half sat. [mgO2/L]	0.05	0.1
Anoxic/anaerobic NOx half sat. [mgN/L]	0.15	0.2
AOB DO half sat. [mgO2/L]	0.25	0.75
NOB DO half sat. [mgO2/L]	0.5	0.75

$$\frac{mg\ TKN}{L} = mg\ NH_3 \times \frac{1\ mmol\ NH_3}{17.031\ mg\ NH_3} \times \frac{1\ mmol\ NH_3\ N}{1\ mmol\ NH_3} \times \frac{14.0067\ mg\ NH_3\ N}{1\ mmol\ NH_3\ N} \times \frac{1\ mg\ TKN}{0.75\ mg\ NH_3\ N}$$

Table C14: Summary of seasonal influent quality showing design (grey) and converted values (white).

	Summer	Winter
BOD (mg BOD/L)	100	140
COD (mg COD/L)	210	294
TSS (mg TSS/L)	113	180
ISS (mg ISS/L)	17	27
NH3 (mg NH3/L)	17	22
TKN (mg TKN/L)	19	24
TP (mg TP/L)	4.0	6.3

Alum use was calculated when phosphorus limits were not achieved, as described in L-RABR design and modeling using the parameters in Table C9. Electricity use for pumping was calculated for using the parameters in Table C15. Electricity for aeration was calculated using the parameters in Table C16.

$$Elec.Use_{pumping}\left[\frac{kWh}{day}\right] = \frac{\rho_f\left[\frac{kg}{m^3}\right] \times g\left[\frac{m}{s^2}\right] \times h_{S,L}[m] \times Q\left[\frac{m^3}{day}\right]}{\eta_h \times \eta_m} \times \frac{day}{86{,}400\ s} \times \frac{1\ kW}{1000\ W} \times \frac{24\ h}{day}$$

ρf = density of fluid pumped, kg/m^3; g = gravitational acceleration = 9.8 m/s^2; $h_{S,L}$ = head losses, static and in pipes, m; Q = flow, m^3/day; η_h = hydraulic efficiency; η_m = motor efficiency

Table C15: Parameters used to calculate energy use for pumping.

Parameter	Value and Units
$h_{S,L}$	5 m
η_h	0.7
η_m	0.9

$$N/N_0 = \left(\frac{\beta C_{walt} - C_L}{9.17}\right) 1.024^{T-20}\alpha$$

N/ N_0 = oxygen transfer correction factor (hp standard conditions/hp field conditions); β = salinity-surface tension correction factor, usually 1 = 1; C_{walt} = oxygen saturation concentration for tap water at given temperature and altitude, mg/L; C_L = operating oxygen concentration, mg/L = 2 mg/L; T = temperature, °C; α = oxygen transfer correction factor for waste, = 0.82 for municipal WW influent

$$Elec.Use_{aeration}\left[\frac{kWh}{day}\right]=\frac{P_{BW}[hp]}{N/N_0\times\eta_m}\times\frac{0.7457\ kW}{hp}\times\frac{24\ h}{day}$$

P_{BW} = total power uptake, as reported by BioWin, hp

Table C16: Parameters used to calculation energy use for aeration.

Parameter	Summer	Winter
Standard O_2 transfer rate, lb O_2/hp·h	3.5	3.5
C_{walt}	8.0	8.9
T, °C	18	13
N/ N_0	0.51	0.52
motor efficiency	0.9	0.9

5 – Volumetric Flow Uncertainty

Volumetric flow data were used to estimate of the influent flow for both L-RABR and BNR-AS scenarios. The gamma distribution for each month was determined by estimating shape and rate parameters (alpha and beta, respectively) based on reported data from each month (Tables C17 and C18). The process of estimating these parameters was to test a range of alpha and beta values, generate random values within the test gamma distribution, and determine the percent of values that occur in three bins, ranging from the minimum to the maximum values found in the data. These bin fractions were compared to the real data, and the process was repeated until no further reduction in the error between real and generated distributions could be attained. The resulting probability density function was used to generate 1000 values of influent flow for each month during MCA (Figure C5).

Table C17: Shape (α, alpha) and rate (β, beta) parameters for each month's gamma distribution of influent flow.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Alpha	3	4	3	2.5	4	4	5	7	7	3	6	2
Beta	9.5	11	9	4	7.5	8.5	8.5	4	8	9	8	10

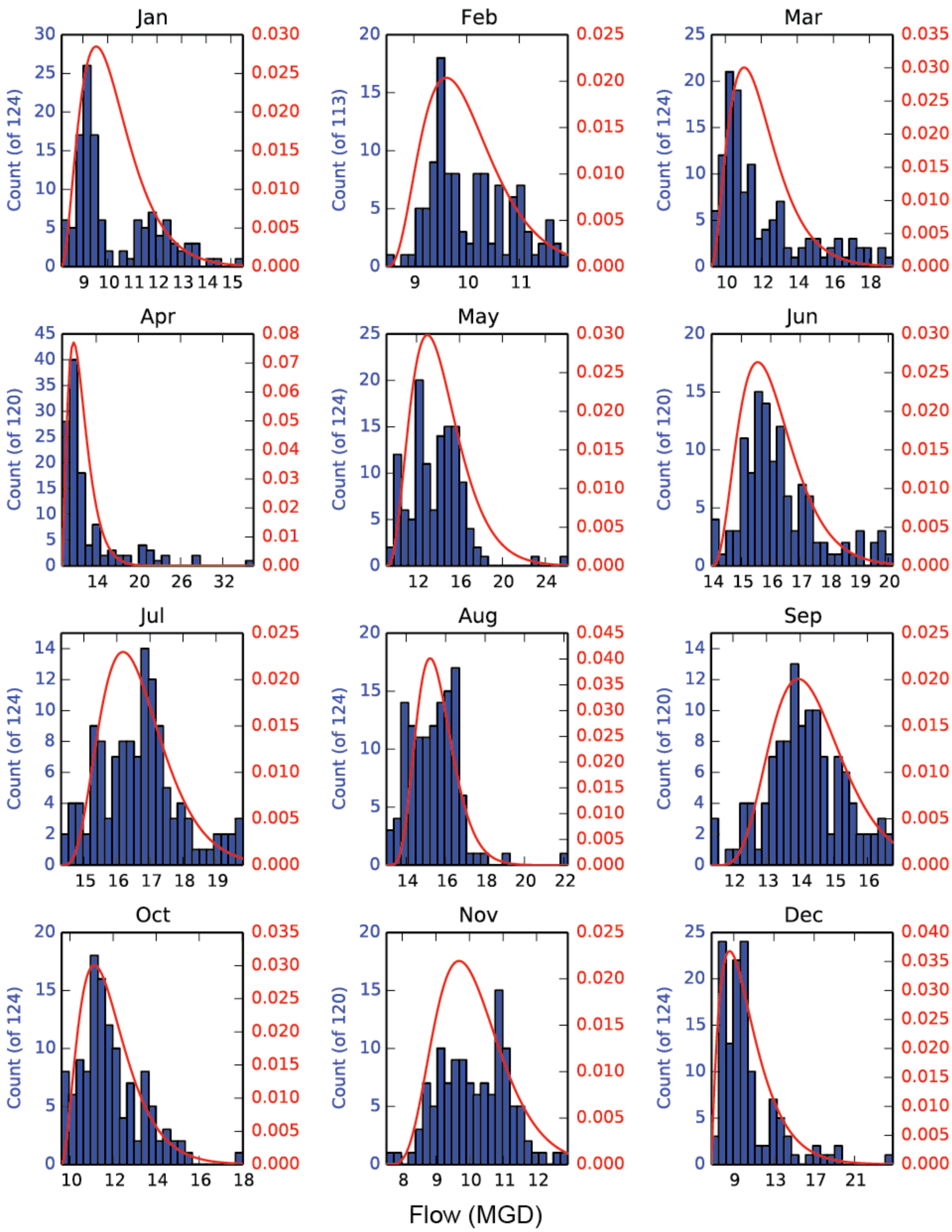


Figure C5: Histograms of volumetric flow data (2010-2013) (blue bars, left axes) and probability density functions (red lines, right axes expressed as probabilities) for each month.

Table C18: Comparison statistics for volumetric flow data (2010-2013) and 1000 values generated for Monte Carlo analysis, reported in million gallons per day (MGD).

	Mean		Median		Min		Max	
	Data	MCD	Data	MCD	Data	MCD	Data	MCD
Jan	10.3	10.3	9.4	10.1	8.1	8.3	15.5	14.8
Feb	10.1	10.0	9.9	9.9	8.5	8.7	11.9	14.4
Mar	12.1	11.9	11.0	11.6	9.2	9.4	19.3	19.9
Apr	12.8	11.9	10.9	11.5	9.2	9.2	36.4	19.8
May	13.8	14.2	13.9	13.8	9.2	9.7	26.1	25.7
Jun	16.3	16.1	16.0	16.0	14.0	14.3	20.1	21.5
Jul	16.7	16.6	16.7	16.5	14.3	14.6	19.8	21.3
Aug	15.4	15.6	15.4	15.4	13.0	13.4	22.2	19.9
Sep	14.1	14.4	14.1	14.3	11.4	11.8	16.7	20.7
Oct	11.9	11.9	11.6	11.6	9.6	9.7	18.0	17.4
Nov	10.1	10.1	10.1	9.9	7.5	7.9	12.9	14.4
Dec	10.4	10.4	9.7	9.9	6.8	6.9	24.8	26.0

6 – L-RABR Influent Quality Uncertainty

Historical data described in the *Historical Lagoon Data* section was used to determine distribution of influent characteristics for the RABR model during MCA. In addition to four years of effluent monitoring data, 14 months of nitrogen and phosphorus data for lagoon pond D (where the RABR system is designed) were also provided, overlapping the 4 years of effluent data. The data were compared for the given time period (Figure C6) and it was observed that the concentrations of ammonia and phosphorus are fairly consistent between pond D and the effluent. Therefore, the effluent values were used to estimate conditions in pond D.

Due to a lower sampling frequency for water quality parameters than for volumetric flow, there was not enough data to determine the expected distribution for each month. Therefore, all of the data for each of the parameters BOD, TSS, and TP were used to estimate alpha and beta for each parameter's gamma distributions (Tables C19 and C20); the probability density function of each gamma distribution was scaled to the range observed for that month to generate 1000 values during MCA (Figures C7-C10). The minimum detection limit (2.5 mg/L) was reported in the data for all BOD values below that threshold. Therefore, these values were excluded when generating the gamma distribution; when the final 1000 values used for MCA were generated, 74.7% came from the gamma distribution and 25.3% were randomly distributed between 0 and 2.5 as 25.3% of data samples were 2.5 mg/L (Figure C7).

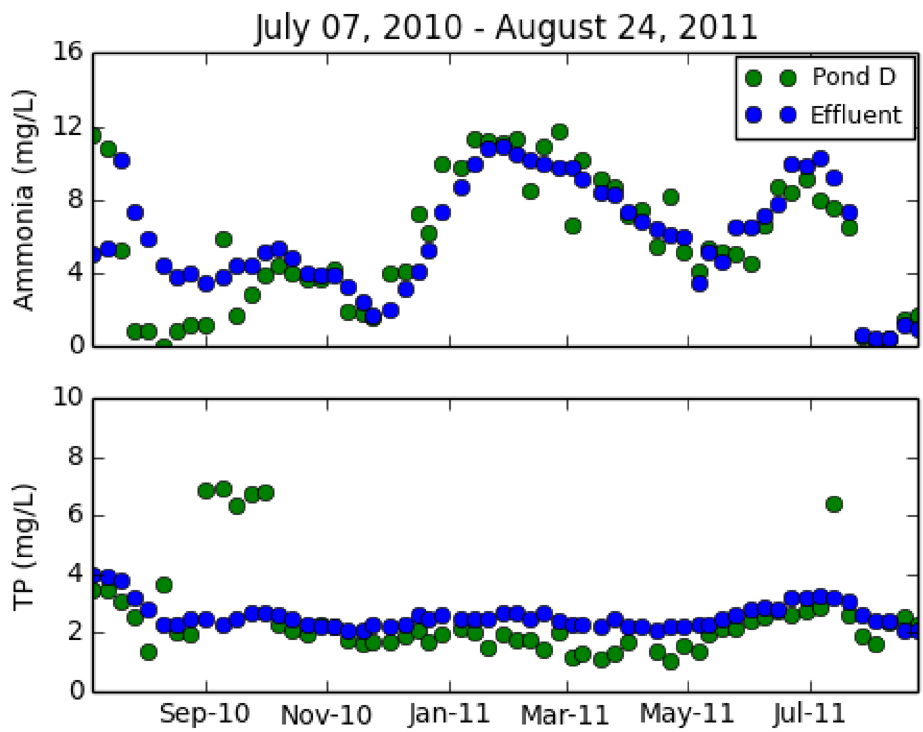


Figure C6: Comparison of ammonia and phosphorus levels in lagoon Pond D and effluent for all available Pond D data.

Ammonia showed a stronger seasonal effect than other quality parameters, with significantly different values and distributions of data for summer (May – October) and winter (November – April). As a result, two normal distributions were used. The resulting normal distributions generated unrealistic (negative) ammonia concentrations. Because of the high number of summer concentrations between 0 and 1 mg/L, a random value was generated in that range to replace the negative ammonia concentrations. For the winter months which had no values in such a low range, values were continuously generated within the normal distribution until 1000 positive values were recorded.

Table C19: Shape (α, alpha) and rate (β, beta) parameters for BOD, TSS, and TP gamma distributions.

	BOD	TSS	TP
Alpha	4	5	3
Beta	8.5	6	8

Table C20: Comparison of statistics for BOD, TSS, TP, and NH3 data (2010-2013) and 1000 values generated for MCA, reported in milligrams per liter (mg/L).

	Mean		Median		Min		Max	
	Data	MCA	Data	MCA	Data	MCA	Data	MCA
BOD Gamma	20.3	19.4	19.0	18.4	6.0	7.4	45.0	45.2
BOD All	15.8	14.8	16.0	15.6	2.5	0.0	45.0	45.2
TSS	25.5	25.6	22.0	23.5	2.0	4.7	80.0	68.9
TP	3.0	3.0	2.9	2.8	1.9	2.0	6.2	6.6
	Mean		**St. Dev.**		**Min**		**Max**	
	Data	**MCA**	**Data**	**MCA**	**Data**	**MCA**	**Data**	**MCA**
NH3 Summer	5.3	5.3	3.7	3.4	0.1	0.0	14.4	16.6
NH3 Winter	11.7	11.7	4.8	4.8	1.7	0.0	22.1	26.1

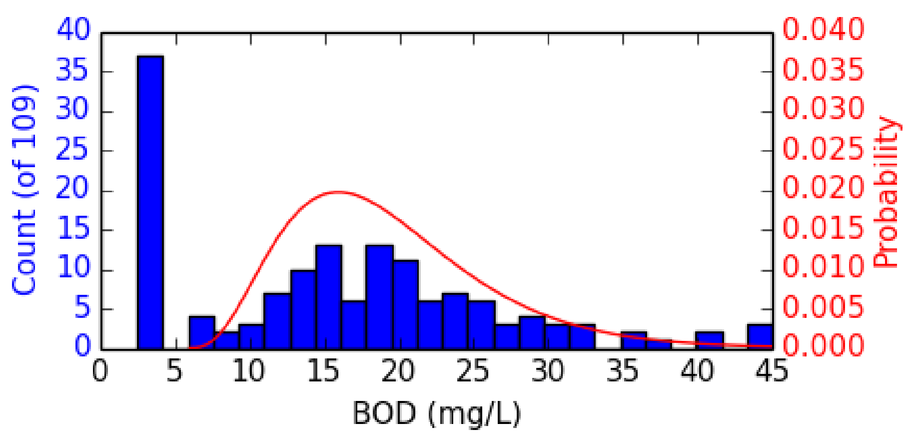


Figure C7: Histogram of BOD data (2010-2013) (blue bars) and probability density functions (red line). Minimum detection values reported (2.5 mg/L) were not considered in the gamma distribution.

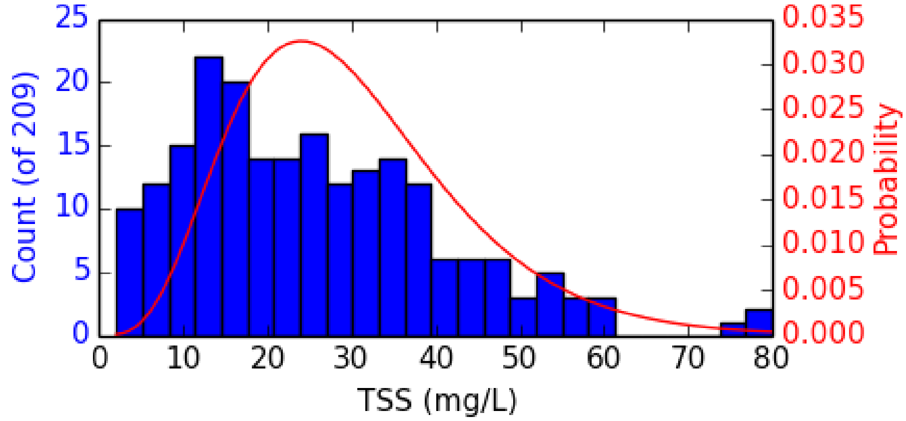


Figure C8: Histogram of TSS data (2010-2013) (blue bars) and probability density functions (red line).

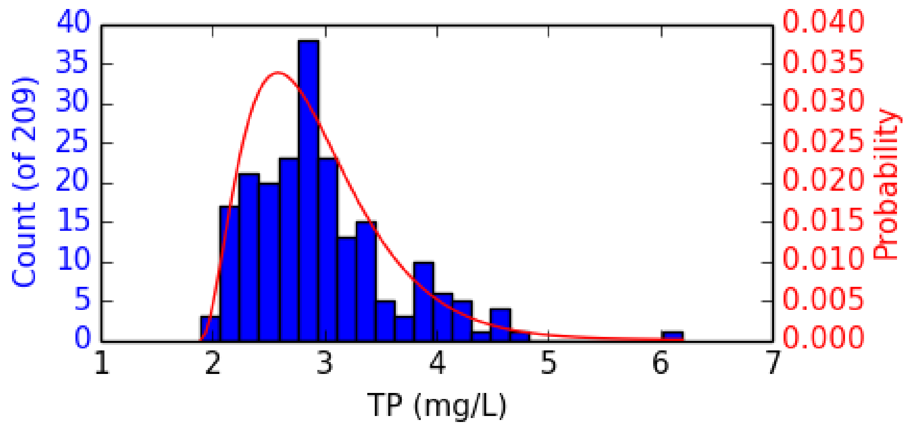
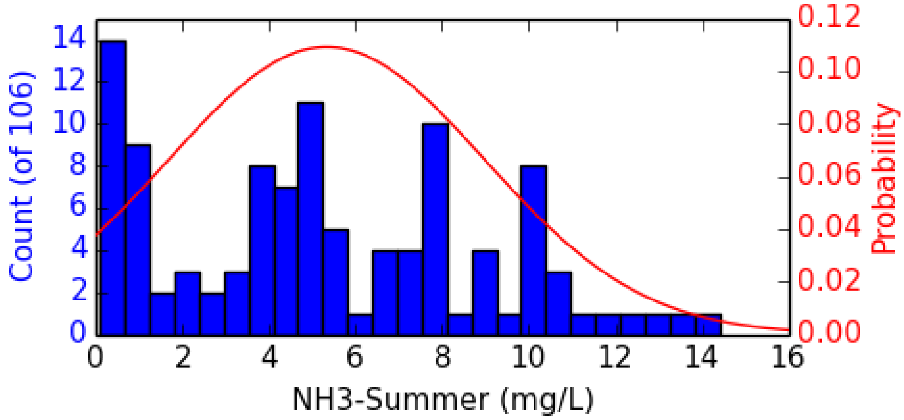


Figure C9: Histogram of TP data (2010-2013) (blue bars) and probability density functions (red line).



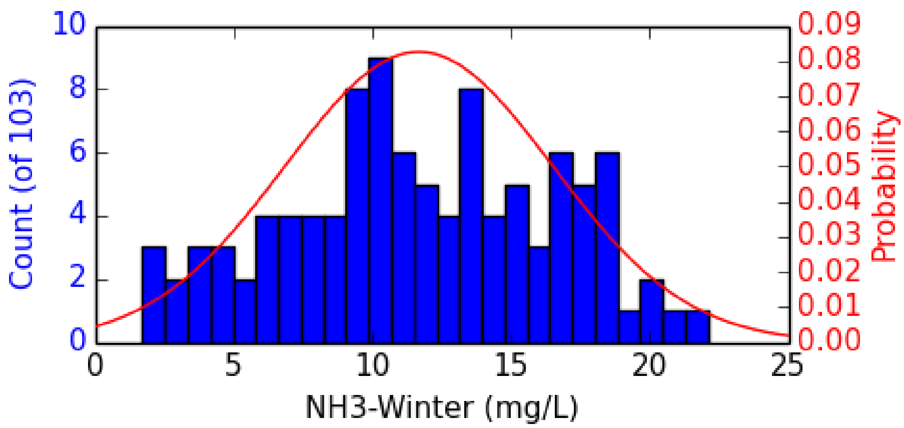


Figure C10: Histograms of summer (top) and winter (bottom) NH_3 data (2010-2013) (blue bars) and probability density functions (red lines).

7 – BNR-AS Influent Quality Uncertainty

No data was recorded for influent quality to the Logan treatment plant. Average values were instead estimated using design values from the wastewater treatment master plan for the city of Logan (Carollo Engineers, 2013). MCA was also performed for the BNR-AS scenario using 1000 influent conditions for each month.

Since effluent phosphorus is often a function of the influent P concentration, particularly for lagoon systems, the influent TP values were generated using the same gamma distribution created from effluent data and scaled to reflect estimated influent concentrations. BOD, TSS, and N were assumed to be normally distributed. Effluent data from the Logan lagoons were used to estimate the "standard deviations" necessary for generating normal distributions of influent quality for MCA. The estimation method depends on if the effluent data was fit to a gamma distribution† (BOD and TSS) or normal distribution‡ (NH3).

$$StDev_{Design}^{\dagger} = \frac{Max_{Data,GD} - Min_{Data,GD}}{6} \times \frac{Mean_{Design}}{Mean_{Data,GD}}$$

$$StDev_{Design}^{\ddagger} = \frac{StDev_{Data,ND}}{Mean_{Data,ND}} \times Mean_{Design}$$

When performing MCA, it was necessary to perform checks to ensure the consistency of various influent parameters in order for BioWin to accept the influent conditions generated. For COD, if the total COD was less than the theoretical COD from the TSS for that MC iteration, the total COD was increased to equal the theoretical COD from TSS. For nitrogen, if TKN was less than theoretical N from TSS, TKN was increased to equal N from TSS.

$$\frac{COD_{TSS}}{L} = \frac{mg\ TSS}{L} \times \frac{0.85\ mg\ VSS}{1\ mg\ TSS} \times \frac{1.42\ mg\ COD}{mg\ VSS}$$

$$\frac{mg\ TKN_{TSS}}{L} = \frac{mg\ COD_{Total}}{L}$$

$$\times \frac{\left(\frac{0.08\ mg\ COD}{mg\ COD_{Total}} \times \frac{0.035\ mg\ N}{mg\ COD}\right)^{\dagger} + \left(\frac{0.0108\ mg\ COD}{mg\ COD_{Total}} \times \frac{0.086\ mg\ N}{mg\ COD}\right)^{\ddagger}}{1 - \left(\frac{0.75\ mg\ NH_3\ N}{mg\ TKN}\right)^{*} - \left(\frac{0.02\ mg\ N}{mg\ TKN}\right)^{\diamond}}$$

†Unbiodegradable, particulate N; ‡Biomass N; *Ammonia fraction; ◊Soluble, unbiodegradable TKN fraction

Table C21: Summer and winter influent quality statistics of MCA values, reported in milligrams per liter (mg/L).

	COD		TSS		TN		TP	
	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer
Mean	311.2	216.6	99.6	67.8	24	19.2	6.3	4
St. Dev.	78.8	58.3	22.1	15.4	9.5	11.1	1.3	0.8
Median	305.2	212.6	98.2	68.2	24.1	18.6	6	3.8
Min	97.1	72.8	34.8	26.1	2.6	1.7	4.2	2.7
Max	581.9	421.3	168.5	116.3	54.1	57	14	8.9
Count	1000	1000	1000	1000	1000	1000	1000	1000

8 – Operation Stage Inventory

Eutrophication potential was determined from discharge of COD, TN, and TP from all scenarios and from background impacts from alum production and electricity production (EcoInvent v2.2).

Table C22: Conversion factors used in calculating eutrophication potential (EUT)

Conversion Factor	Value	Units
EUT from COD	0.05	kg N-eq/kg COD discharged
EUT from TN	0.986	kg N-eq/kg TN discharged
EUT from TP	7.29	kg N-eq/kg TP discharged
EUT from alum production	3.523e-3	kg N-eq/L alum
EUT from electricity generation	3.342e-3	kg N-eq/kWh

Table C23: Conversion factors used in calculating global warming potential (GWP)

Conversion Factor	Value	Units
GWP from methane	25	kg CO_2-eq/kg CH_4
GWP from nitrous oxide	298	kg CO_2-eq/ kg N_2O
Methane, from lagoons	0.00125	kg CH_4/kg COD removed
Nitrous oxide, from denitrification	0.01	kg N_2O-N/kg N denitrified
Methane, from effluent COD	0.025	kg CH_4/kg COD discharged
Nitrous oxide, from effluent N	0.0025	kg N_2O-N/kg TN discharged
GWP from alum production	0.5907	kg CO_2-eq/L alum
GWP from electricity generation	0.8364	kg CO_2-eq/kWh

Methane emissions which arise from COD removal in anaerobic zones of lagoons (Godin et al., 2012) were included in direct emissions from extant lagoons and the L-RABR scenario. Methane emissions from the BNR-AS scenario were reported in BioWin as a fraction of off-gas from bioreactors and converted to mass flows using dry off-gas flow rates, also reported in BioWin. Nitrous oxide emissions which result from nitrification/denitrification cycles in activated sludge were considered for the BNR-AS scenario and calculated based on denitrification rates reported in BioWin and conversion factor reported previously (Foley et al., 2010). Nitrification/denitrification are not considered a major pathway for nitrogen removal in lagoon systems (Middlebrooks et al., 1999) and thus direct nitrous oxide emissions were not considered for the extant lagoon or L-RABR scenario. Methane emissions from landfilled sludge were considered and calculated based on the EPA's GHG Reporting Rule considering factors provided for sewage sludge and assuming landfilled solids originally have a solids content of 10%.

Conversion factors for EUT, GWP, and CED by alum production and electricity generation were determined using the EcoInvent database processes "RER: aluminium sulphate, powder, at plant" and "US: electricity, low voltage, at grid". Alum was converted to a per liter basis using data in Table C9.

$$G_{CH4} = \left[\sum_{x=S}^{T-1} \left\{ W_x L_{0,x} \left(e^{-k(T-x-1)} - e^{-k(T-x)} \right) \right\} \right]$$

G_{CH4} = Modeled methane generation rate in reporting year T (metric tons CH4); x = Year in which waste was disposed; S = Start year of calculation; T = Reporting year for which emissions are calculated; W_x = Quantity of waste disposed in the landfill in year X ; $L_{0,x}$ = CH4 generation potential (metric tons CH4/metric ton waste) = MCF × DOC × DOCF × F × 16/12; MCF = Methane correction factor (fraction); default is 1;DOC = Degradable organic carbon; DOC (sewage sludge) = 0.05, Weight fraction, wet basis; DOCF = Fraction of DOC dissimilated (fraction); default is 0.5; F = Fraction by volume of CH4 in landfill gas from measurement data, if available (fraction); default is 0.5; k (sewage sludge) = 0.06 to 0.185, yr^{-1}

Table C24: Conversion factors used in calculating global cumulative energy demand (CED)

Conversion Factor	Value	Units
CED from alum production	2.977	kWh/kWh
CED from electricity generation	3.723	kWh/kWh

9 – Construction Phase Inventory

Information for construction stage materials, including EcoInvent processes used, masses considered, and assumptions for both the L-RABR and BNR-AS scenario can be found in Tables C25 and C26. For the L-RABR scenario, these values constitute the equivalent number of RABR units used at the maximum number of stages modeled. For both scenarios, impact results were normalized per year based on 20 years, as designated in the functional unit.

Table C25: Materials and assumptions used in construction stage inventory for the L-RABR scenario

Database Process	Reported group	Assumption	Mass, kg
GLO: yarn, cotton, at plant	Cotton yarn	36.3 kg/wheel, changed yearly	1.6E+07
RER: packaging box production unit	Cotton yarn	*	
CH: disposal, paper, 11.2% water, to sanitary landfill	Cotton yarn	*	
RER: transport, lorry 16-32t, EURO3	Cotton yarn	*	
US: cotton fibres, at farm	Cotton yarn	All US grown cotton used	
US: electricity, low voltage, at grid	Cotton yarn	All US electricity used	
Motor Assembly		1 motor/RABR wheel	
RER: aluminium, production mix, at plant		4.1 kg/motor	
RER: aluminium, primary, at plant	Virgin Aluminum	68% virgin**	6.1E+04
RER: aluminium, secondary, from new scrap, at plant	Recycled Aluminum	32% recycled**	1.9E+04
RER: aluminium, secondary, from old scrap, at plant	Recycled Aluminum	32% of recycled from old scrap**	9.3E+03
RER: steel, electric, un- and low-alloyed, at plant	Steel	6.8 kg/motor	1.5E+05
Pond channel wall construction		3 walls into pond D	
CH: concrete, normal, at plant	Concrete	600 m^3/wall, 2380 kg/m^3**	4.3E+06
RER: reinforcing steel, at plant	Steel	77.58 kg/m^3 concrete***	1.4E+05
RABR wheel assembly			
RER: aluminium, production mix, at plant		54 kg/RABR wheel	
RER: aluminium, primary, at plant	Virgin Aluminum	68% virgin**	8.0E+05
RER: aluminium, secondary, from new scrap, at plant	Recycled Aluminum	32% recycled**	2.6E+05
RER: aluminium, secondary, from old scrap, at plant	Recycled Aluminum	32% of recycled from old scrap**	1.2E+05
RER: steel, electric, un- and low-alloyed, at plant	Steel	4.6 kg/RABR wheel	1.0E+05

*Default values for EcoInvent process "GLO: yarn, cotton, at plant"

**From EcoInvent documentation for process "RER: aluminium, production mix, at plant"

***From (Doka 2003) as reported in (Foley 2010)

Table C26: Materials and assumptions used in construction stage inventory for the BNR-AS scenario

Database Process	Reported group	Assumption	Mass, kg
CH: bitumen, at refinery	Asphalt	0.5 kg/m^3 concrete*	3.86E+03
CH: concrete, exacting, at plant	Concrete	2380 kg/m^3**	1.84E+07
CH: limestone, crushed, washed	Limestone	21.45 kg/m^3 concrete*	1.66E+05
CH: rock wool, packed, at plant	Other	0.87 kg/m^3 concrete*	6.72E+03
GLO: chemicals inorganic, at plant	Chemicals	0.5 kg/m^3 concrete*	3.86E+03
GLO: chemicals organic, at plant	Chemicals	4.05 kg/m^3 concrete*	3.13E+04
RER: aluminium, production mix, at plant		0.87 kg/m^3 concrete*	
RER: aluminium, primary, at plant	Virgin aluminum	68% virgin**	4.57E+03
RER: aluminium, secondary, from new scrap, at plant	Recycled aluminum	32% recycled**	1.45E+03
RER: aluminium, secondary, from old scrap, at plant	Recycled aluminum	32% of recycled from old scrap**	6.99E+02
RER: chromium steel 18/8, at plant	Steel	6.23 kg/m^3 concrete*	4.81E+04
RER: glass fibre, at plant	Other	1.96 kg/m^3 concrete*	1.51E+04
RER: polyethylene terephthalate, granulate, amorphous, at	Plastic	2.46 kg/m^3 concrete *	1.90E+04
RER: polyethylene, HDPE, granulate, at plant	Plastic	2.44 kg/m^3 concrete*	1.89E+04
RER: polyethylene, LDPE, granulate, at plant	Plastic	0.02 kg/m^3 concrete*	1.55E+02
RER: reinforcing steel, at plant	Steel	77.58 kg/m^3 concrete*	5.99E+05
RER: synthetic rubber, at plant	Other	0.88 kg/m^3 concrete*	6.80E+03
RNA: copper, primary, at refinery	Copper	0.92 kg/m^3 concrete*	7.11E+03

*From (Doka, 2003) as reported in (Foley et al., 2010)

**From EcoInvent documentation for process "RER: aluminium, production mix, at plant"

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